United States Department of Agriculture

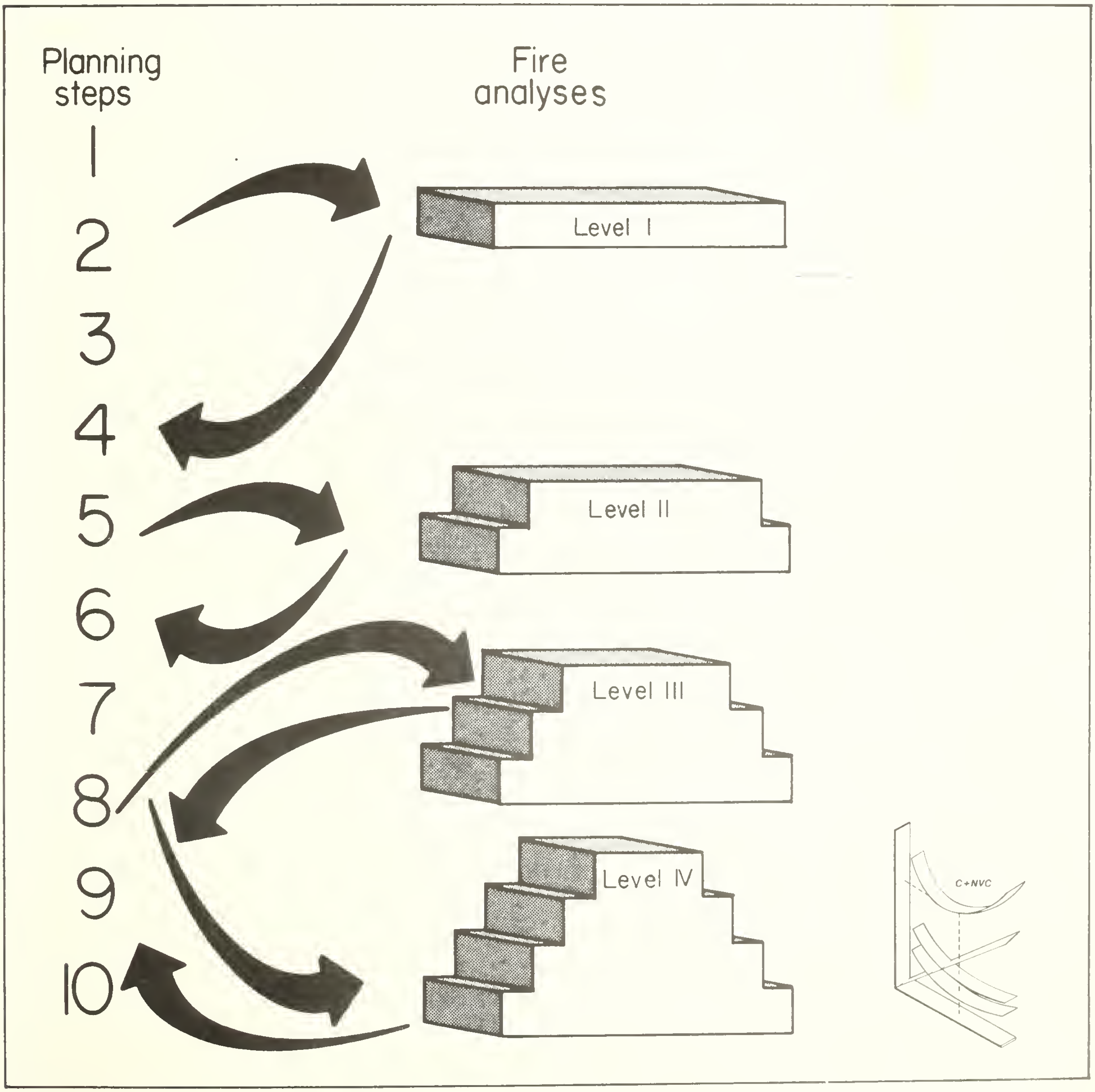
Forest Service

Pacific Southwest Forest and Range Experiment Station

General Technical Report PSW-74

Integrating Fire Management Analysis into Land Management Planning

Thomas J. Mills



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Mills, Thomas J. **Integrating fire management analysis into land management planning**. Gen. Tech. Rep. PSW-74. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1983. 8 p.

The analysis of alternative fire management programs should be integrated into the land and resource management planning process, but a single fire management analysis model cannot meet all planning needs. Therefore, a set of simulation models that are analytically separate from integrated land management planning models are required. The design of four levels of fire management analysis that contribute to the planning process has been developed. The interactions among these four levels and between the fire management analysis and the planning process are designed for consistency and analysis efficiency. These analytical models emphasize economic efficiency and risk consequences of fire management program options.

Retrieval Terms: economic efficiency, risk, probability modeling, fire suppression.

Cover: To screen fire management options, a ten-step process progressing through four levels of analysis has been designed.

Publisher

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

November 1983

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Analysis of alternative long term natural resource programs pose challenges to both planners and managers. One dimension of the challenge is the diversity of both management actions and resource outputs affected by those actions. Management actions that vary from timber harvesting to road building, and from cover type conversion to fire suppression, for example, affect both market priced outputs, such as timber and range yields, and nonmarket priced outputs, such as water, wildlife, and visual quality. Each of these management actions and outputs have unique aspects that require special analysis, but each must also be addressed in an integrated resource analysis to avoid achieving a high level of one output at the expense of excessive reductions of another output, i.e., to avoid suboptimization.

Another dimension of the challenge is the wide range of temporal and spatial detail, i.e., levels of analysis resolution, that must be considered in long term planning. For example, the initial screening among numerous long term management programs can be completed at a relatively low time and space resolution. It may be sufficient at that stage to evaluate programs for generic "types" of management areas, rather than site-specific areas. Eventually, however, the planning must address a few site-specific management actions in greater detail for real time decisions.

If these challenges are met with a set of interrelated and complementary analytical tools, it will be possible to develop information efficiently for complex resource management decisions. If, on the other hand, the analytical models are not sufficiently interrelated and do not adequately address the question of time and space consideration, the information produced from the separate models is likely to be inconsistent and costly. A common error is the application of highly site-specific models at early stages in the screening among program alternatives. Being highly site-specific is extraneous at that point in the overall evaluation framework.

This report describes a framework for simulation models that integrate fire management analysis into the land planning process. Similar frameworks are appropriate for other functional programs, such as timber and recreation. Examples are drawn from Forest Service, U.S. Department of Agriculture situations, but the same general concepts apply to any organization with wildland fire management responsibilities.

FIRE MANAGEMENT

Major changes in the land management planning process on National Forest lands were formalized in the National Forest Management Act of 1976 and subsequent regulations for its implementation (U.S. Dep. Agric., Forest Serv. 1979a). The Act requires that the planning process fully integrates the various resource components of the management program, and that the criteria for designing and selecting alternatives be explicit.

The revised National Forest fire management policy contains related changes—the fire management program must be cost effective and consistent with land management objectives (U.S. Dep. Agric., Forest Serv. 1978). Minimization of fire program *c*ost plus the *n*et *v*alue *c*hange (C + NVC) in resource outputs has been specified as the cost effectiveness criterion (U.S. Dep. Agric., Forest Serv. 1981a).

Four "levels" of fire management analysis, each one designed to address different questions at appropriate levels of time and space analysis detail, have been described to help implement these policy changes (U.S. Dep. Agric., Forest Serv. 1979b). Analytical models for portions of the four-level framework are described in a handbook (U.S. Dep. Agric., Forest Serv. 1982). The process in the handbook contains major improvements over earlier procedures and is a step forward in the evolution of complete and analytically efficient procedures. A more complete and consistent framework for full fire management program analyses is needed, however.

Although economic efficiency analysis of fire control programs is emphasized here, other fire program activities, such as prescribed burning, and other effects not easily placed within an economic analysis are also important. The ecological effects of fire, such as those documented by Davis and others (1980), Kilgore (1979), and Parsons and DeBenedetti (1979), are also important long term factors that should be considered in the fire program analysis. Those ecological factors might more appropriately be addressed during deliberations of the interdisciplinary planning team. Gorte and Gorte (1979) reviewed the economic efficiency dimension of fire

management program evaluations, and Martell (1982) reviewed operations research application to fire management decisionmaking.

INTEGRATED PLANNING PROCESS

In the past, the management plan for a Forest Service administration unit was sometimes a simple summation of the separate plans prepared for each resource function. This practice often led to suboptimal programs and direct conflicts between functional activities. Fire control before 1978 is an example. The fire planning criterion (U.S. Dep. Agric., Forest Serv. 1972) was to control all fires before they reached a size of 10 acres, and the policy was to take action to achieve control by 10 a.m. of the second day if initital attack on the first day was unsuccessful. This policy sometimes led to aggressive fire suppression which was in direct conflict with resource objectives—especially in low intensity fires.

The National Forest land management planning process (U.S. Dep. Agric., Forest Serv. 1979a) was designed to resolve this type of conflict. The planning process is an interdisciplinary and fully integrated approach to decisionmaking. Issues and concerns are addressed by an intentionally broad array of management alternatives. The alternatives are evaluated against several explicit planning criteria in a process open to public scrutiny. The selected alternative is then converted into an annual operating plan and its implementation is monitored to ensure consistency with the planning criteria and objectives.

A fully integrated evaluation of management alternatives, though a desirable goal, is difficult to achieve analytically. For example, only one of the many potential fire management program options, from a wide array of technically feasible options, can be incorporated into the management alternative during functionally integrated analysis. The included fire program must be consistent with both the objective of the alternative and the planning criteria. If a less-than-optimal fire program is included and its cost is a major share of the total management cost, the management alternative may be rejected—not because it is not the best one, but because it is not configured in its optimal form.

One solution to this problem is to develop the fire management program options for each integrated management alternative in a model that is analytically separable from an integrated analysis model (U.S. Dep. Agric., Forest Serv. 1981b). The costs and resource effects of the selected fire option would be included in the integrated analysis model along with similar estimates from other resource programs. Because the fire management analysis model would be narrower in scope than the integrated analysis model, a more thorough screening of fire program options could be accomplished at a lower cost than under alternative approaches. This approach does *not* advocate a return to single-resource planning. Rather it simply recognizes the analytical complexity of the fire system and the practical limits of the integrated analysis model.

Definitions

For the purposes of this report, the following terms and their definitions are used: A *management objective* is a land and resource condition, and a multiresource output stream, which a management program is designed to achieve. A *management alternative* is a multifunctional land and resource management program of actions formulated to accomplish the management objective. A *management prescription* is a specific management action taken on a particular parcel of land; the sum of various prescriptions describes the management alternative. *Integrated analysis*, as opposed to fire management analysis, is an evaluation of multiresource or multifunctional programs. The *fire program mix* is the composition of inputs, such as initial attack forces versus acres of fuel treatment, purchased with the budget specified by the program level. The *fire program option* is a particular combination of dollar program level and program mix. The *management area* is the land unit to which a particular fire program option is applied.

Premises

Three premises underlie the design of this fire management analysis framework.

Premise 1: Fire management programs do not simply *support* resource management activities; they *contribute* directly to the accomplishment of resource management objectives by affecting resource output levels and program costs. They should, therefore, be considered as options for addressing the issues and concerns at the same time as other resource management opportunities.

Premise 2: Evaluation of fire management program options is so complex that separate analytical models are needed even if some suboptimization results. The suboptimization is probably far less than the cost of the errant decisions which result from an insufficient screening of fire program options within the main body of integrated planning models.

Premise 3: A single analytical model cannot efficiently perform at all the required levels of time and space detail. A universal fire program analysis model is infeasible, or at least very inefficient. The fire management analysis must also be clearly linked to and guided by the preliminary decisions generated within the main body of integrated planning.

Levels of Analysis

Four levels of fire management analysis are proposed to accomplish the screening of fire program options. These levels

and their objectives vary somewhat from those described elsewhere, (U.S. Dep. Agric., Forest Serv. 1979b), particularly in the use of acreage-burn standards or pars.

Level I

Level I fire management analysis provides a prescreen of fire program options that contribute most toward the accomplishment of the tentative management alternative (*fig. 1*). Since both fire management program level and program mix affects program performance, discrete fire program options which address both the level and mix should be evaluated. The low resolution Level I analysis deals with the time and space dimensions of the analysis in a relatively shallow, but broad manner.

Three types of input information are required: (a) a list of tentatively stated management alternatives as derived from the issues, concerns, and opportunities identified in Step 1 of the integrated planning process, expressed in terms of an intended time stream of resource outputs; (b) the planning criteria (Step 2); and (c) a description of a representative management area to which each tentative management alternative applies, including parameters that influence fire program performance and effects, such as resource values, terrain, fire occurrence, and vegetation type (Step 3).

The revised National Forest fire management policy (U.S. Dep. Agric., Forest Serv. 1981a) stresses the importance of economic efficiency as a planning criterion. This importance had been underscored earlier by the U.S. Office of Management and Budget and the U.S. Senate (1978) in their inquiries about the Forest Service's fire programs (U.S. Dep. Agric., Forest Serv. 1977). The minimization of the sum of the program cost plus the fire-induced net value change in resource outputs and improvements is an appropriate economic efficiency criterion for fire programs (Gorte and Gorte 1979, Mills 1979, Simard 1976). Quantitative effects of fire on resource outputs is another criterion even though some effects are already included in the C + NVC calculation. Risk consequences is another important criterion in Level I analysis.

Several pieces of output information are produced from Level I for each tentative management alternative and fire program option evaluated on the representative management area. One output is the expected C + NVC. The expected value is a probability weighted average. Another output is the expected net change in the output of resources. The net change is the output levels without fires minus the output level that would occur with the fires simulated to occur under the fire program option being evaluated. A third output is the risk consequences as measured by the shape of the cummulative probability distributions about C + NVC and resource output change. The last output is the expected value number of fires or acres burned or both by fire size and fire intensity classes which is the basis for a planning criterion in Level III.

Selection of a fire program is determined by the relative weights or constraint levels or both applied to the planning criteria. If the decisionmaker does not specify the relative weights in advance, the minimum expected value C + NVC should be used as a default criterion since it does the most complete job of incorporating program costs, resource outputs and values, and the stochastic nature of the fire management program.

The fire program cost and resource effect coefficients thus derived are incorporated into the integrated analysis of the management situation (Step 4). The integrated alternatives are

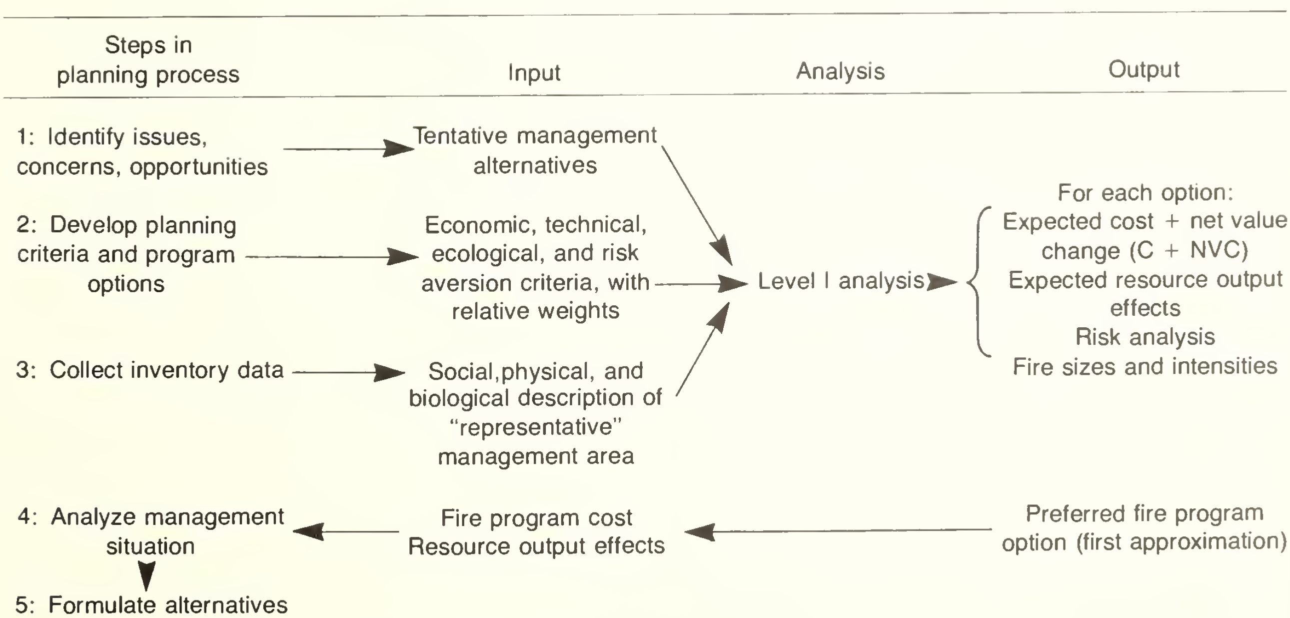

Figure 1—In Level I of the fire management analysis, a broad range of fire program options is prescreened to determine which one contributes the most to the tentative land management alternative.

then formulated (Step 5) with the benefit of this fire program prescreening.

The Fire Economics Evaluation System (FEES) now being developed by the Forest Service will meet the design requirements for Level I (Bratten 1982, Mills and Bratten 1982). The system will evaluate widely different fire management programs applied to management areas that are described in situation-specific, but not site-specific, terms. This design is built on the premise that there are classes of management areas that exhibit essentially the same fire program performance for this first level screening.

Level II

Guided by the formulated alternatives (Step 5) and the fire program selected in Level I, the Level II model performs a more detailed analysis on a more narrowly defined range of fire program options, thus refining the Level I fire program selection (*fig. 2*). The refined cost and resource output change coefficients are incorporated into each integrated alternative (Step 6). The planning process then moves through the evaluation of management alternatives (Step 7) and the selection of one alternative (Step 8).

Analysis at this level is characterized by increased specificity of site and time dimensions of the analysis and concentration on a narrower range of program options. This narrowing of analysis scope is a major source of analysis efficiency. Without a Level I analysis, the Level II analysis would probably be applied to a far too narrow range of program options because the cost of evaluating each program option is greater.

Input information required in Level II is similar to that in Level I: a description of planning criteria, a site-specific description of the management area, and the formulated management alternatives in terms of intended resource outputs. Risk is analyzed by a probabilistic model in Level I, so Level II can be restricted to expected value results. Additional criteria include site-specific fire constraints. For example, fires in an area containing cultural resources may be restricted to an average of no more than 10 acres.

Four categories of output are developed from the Level II analysis: (a) the expected value C + NVC; (b) the fire-induced expected value resource output change; (c) whether the fire constraint is satisfied; and (d) the expected number of fires, by fire size and fire intensity. The opportunity cost of the fire constraints must be evaluated carefully, at least by showing the foregone C + NVC.

The Level II results for fire program costs, resource output effects, and number of fires by size and intensity should be similar to those estimates from the Level I analysis for the same fire program option. If they are not consistent, the analysis and/or data resolution in Level I may have been too gross, or the tentative management alternatives in Level I could be much different than the formulated management alternatives analyzed in Level II. If the Level I-II difference is small, Level II will do just what it is designed to do, refine the Level I estimates. If the difference is substantial, it may be more efficient to repeat the Level I prescreening for the formu-

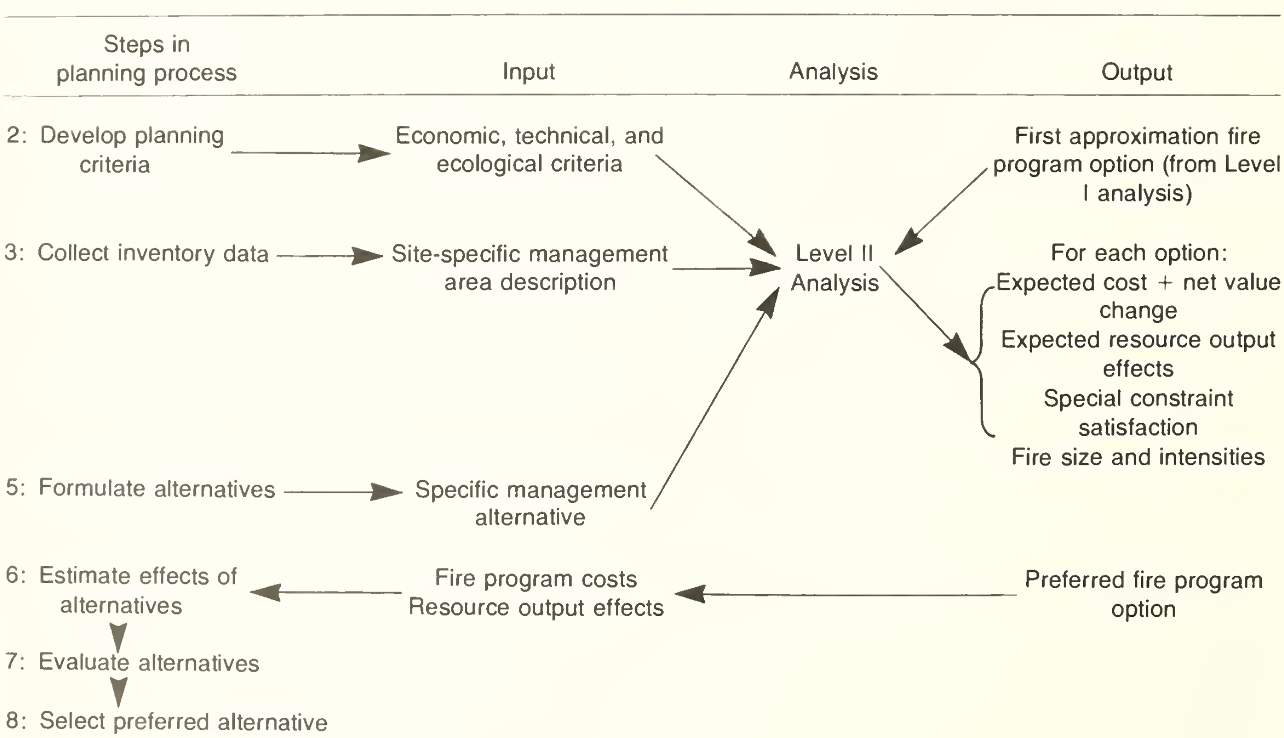

Figure 2—In Level II of the fire management analysis, the first approximation of preferred option derived in Level I is refined.

lated alternatives than to apply the Level II analysis on a large number of options.

The Forest Service's (1982) Fire Management Analysis and Planning Handbook (FSH 5109.19) describes an operational system that provides most of the analysis capability required by this Level II design. The analysis detail is properly greater than in the Level I analysis but it still falls short of the site-specificity needed in later analyses. The Handbook evaluates fire program performance at "representative" fire locations, for example, rather than at all possible fire locations. The representative location structure permits consideration of the fire site constraints, but avoids the analysis cost of greater specification than is needed at this point.

Level III

The Level III fire management analysis transforms the fire program option embodied within the selected management alternative (Step 9) into an annual, operational or implementation plan for the fire management program (*fig. 3*).

Four categories of Level III input are required. One is a description of the management area, but a more site- and time-specific description than in Level I and Level II. Level III data, for example, may include a travel-time network whereas access in Level I may be represented through historical distributions of initial attack arrival times (Mees 1983). The second input is the management direction implied in the selected management alternative. The management direction in Level III is represented by a par or standard for acres burned by fire size and fire intensity derived from Level II output. The use of pars leads to a reduction in Level III analysis complexity since the fire effects and resource values estimates required for the C + NVC evaluation are not needed. The pars may be augmented by the special fire constraints, the third input. Since the pars were guided in Level II by economic efficiency, resource output, and risk criteria, those criteria can be replaced in Level III with a cost minimization criterion. The fourth input is the Level II fire program option contained in the preferred alternative. This limits the range of program options evaluated in Level III considerably.

The development and use of pars advocated here is different than what has been proposed elsewhere (U.S. Dept. Agric., Forest Service 1979b). Attempts to develop pars directly from the integrated planning process had serious shortcomings since it is difficult to transform the resource outputs of each management alternative directly into pars. Too much weight was given to the historical acreage burned. That effort attempted to establish pars as absolute acre-burn ceilings, levels above which intolerable losses occur. An "intolerable loss" from fire is not meaningful, if intolerable implies infinite disbenefit. Furthermore, no place to judge the reasonableness of the cost of achieving the pars was provided for. The use of pars proposed here overcomes these difficulties because the pars are developed in Levels I and II before they are used as a selection criterion in Level III. The pars are, therefore, developed after a full consideration of all planning criteria.

Three categories of output are derived for each fire program option evaluated in Level III: (a) an estimate of acres burned by fire size and fire intensity class for comparison with the pars; (b) an evaluation of whether the special constraints are met; and (c) an estimate of the fire program cost.

A major component of the Level III fire plan is an annual budget request. The decisionmaker may question what would result from a budget increase or reduction. If consistency has been achieved, the fire program options evaluated in Level III are a subset of those evaluated at Levels II or I at a less site- and time-specific level of resolution. Estimates of the economic efficiency, resource output, and risk impacts from budget changes can, therefore, be drawn from the results of Levels I or II or both. That information does not have to be included directly in the Level III model.

The FOCUS simulation model (Bratten and others 1981) operates at the resolution appropriate for Level III analysis.

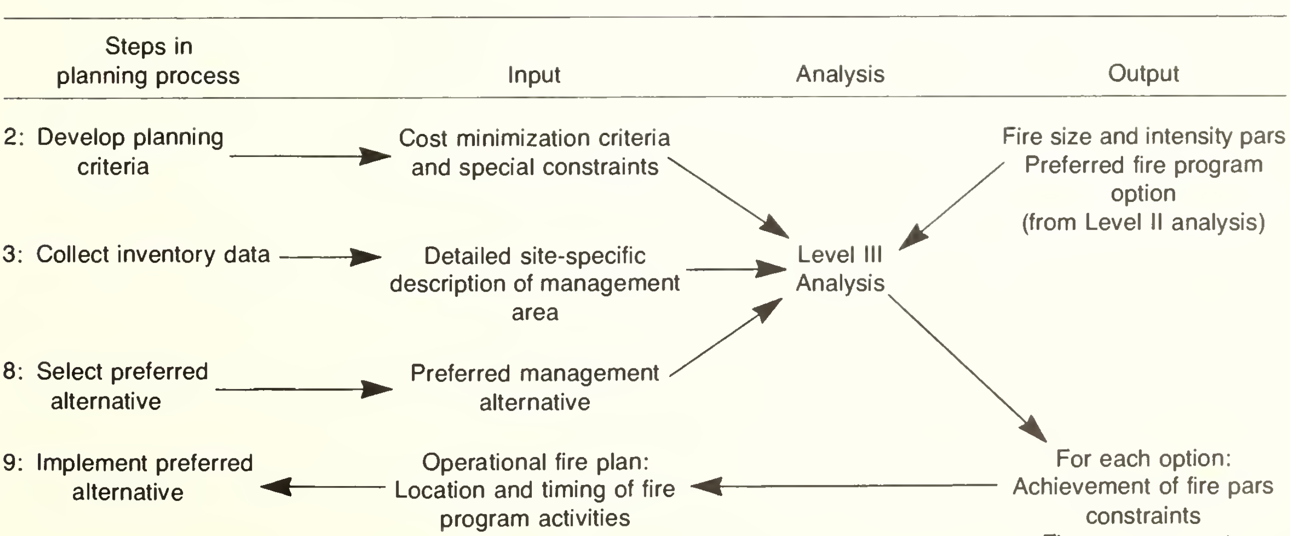

Figure 3—The Level III analysis provides an annual operating plan for the fire program.

The spatial dimension is incorporated by a complete transportation network and a full list of fire locations, either historical locations or potential locations. FOCUS does not provide fire-induced resource output changes or resource values but it does yield estimates of fire sizes and suppression costs which are needed in Level III.

Level IV

The Level IV analysis evaluates program options for individual fire events and develops an action plan for the one option most consistent with the management objective (*fig. 4*). Some of the Level IV analysis may be done in advance of the fire, such as prescribed fire project plans, while others must be completed in real-time, such as escaped fire situation analyses. The following discussion concentrates on escaped fire situation analyses, although similar principles apply to other Level IV analyses.

The output from previous levels provides first approximations of Level IV inputs which are then adjusted in light of the site- and time-specific description of the fire. One input is the expected resource output for the area that the fire may burn, another is the planning criteria (economic efficiency and risk). Additional criteria are special constraints and public safety. Next is a time-specific description of the fire location. Spatial variation and juxtaposition are important here. Three or four future weather time streams and their probabilities are also identified.

The fire suppression options are evaluated for each weather time stream and then weighted together by their respective probabilities to yield an expected outcome. Risk is displayed through the probabilities of each weather pattern and their associated consequences.

The first Level IV output is C + NVC. Second is the expected resource output effects by resource category. Third is the effect on special contstaints and public safety. Fourth is the risk depicted by the consequences of weather variations. Consistency is more important in Level IV than in earlier levels because first approximations of Level IV input data comes directly from prior levels.

Instructions for preparation of an escaped fire situation analysis are contained in the Forest Service Manual (FSM 5130.3) (U.S. Dep. Agric., Forest Serv. 1981a). Seaver and others (1983) describe further advances in the analysis of alternative escaped fire strategies that are possible.

COMPARISON OF ANALYSIS LEVELS

Similarities

In all levels, analysis begins with a statement of the integrated management objective, even though the form of the objective differs: tentative management alternative (Level I), formulated alternative (Level II), or preferred management alternative (Levels II and IV). Several fire program options are evaluated in each level against several planning and decision criteria. Maximizing economic efficiency and minimizing the

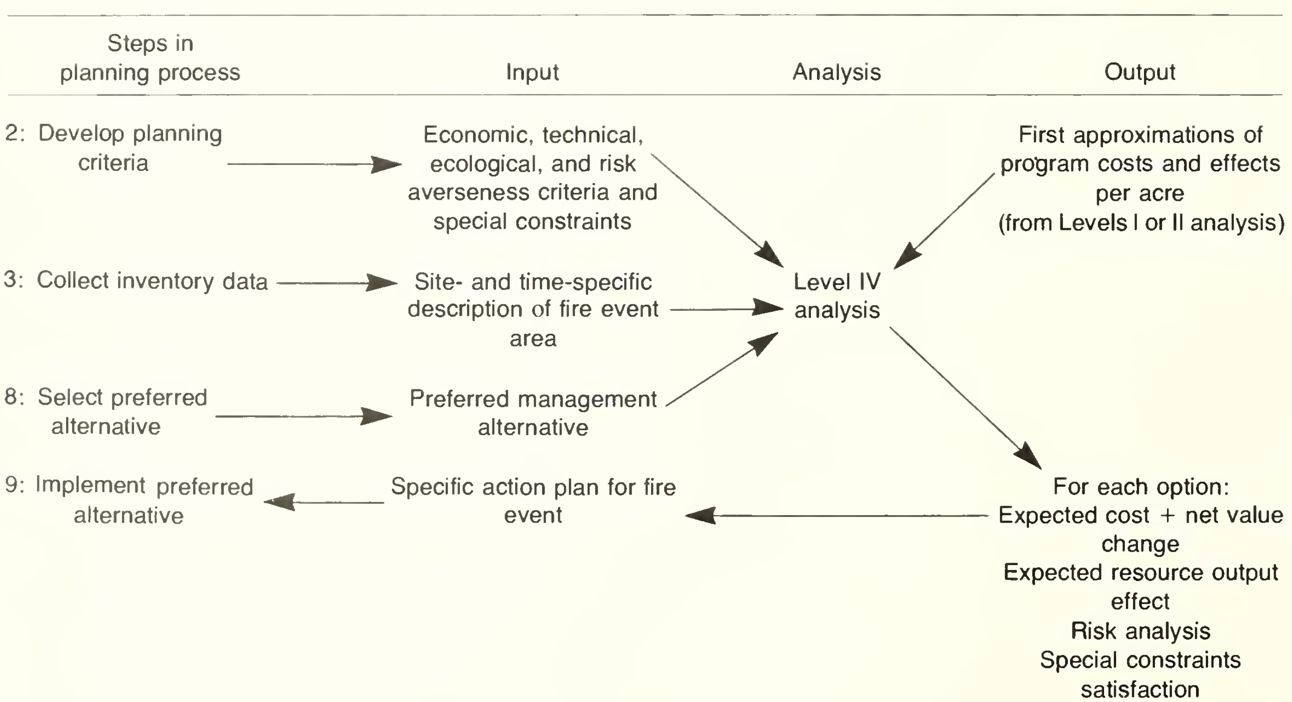

Figure 4—Level IV of the fire management analysis provides an action plan for an individual fire event.

net detrimental effect of fires on resource outputs are common selection criteria in all levels except III, where they are replaced by acre burn pars.

The stochastic impact of fire weather variations is included in all levels and fire occurrence variability is considered in Levels I-III. This variation is reflected through expected value calculations (Levels II and III) or as a probability distribution which accompanies the expected value (Level I and IV).

Differences

Some of the planning criteria differ among the levels. Site-specific, special constraints, such as the exclusion of fire from areas with particularly fragile soils, are considered in Levels II, III, and IV, but are excluded from Level I. The full display of probability outcome is only derived in Level I because probability models require extensive data and are difficult to build. The fullest risk consideration is in Level I so that conclusions reached there can guide analyses in Levels II-IV.

The four levels differ in the breadth of fire program options evaluated. The broad range of options considered in Level I are condensed to a few for the detailed Level II analysis. Level II yields an even narrower range for evaluation in Level III. Different questions are addressed at each level by evaluating a successively narrower set of fire program options.

A progressive increase in temporal and spatial resolution occurs from Levels I to IV. Level I does not require a site-specific area description in the locational sense of juxtaposition of separate parcels of land. Situation-specific "kinds" of fire management situations are evaluated instead in Level I to permit easier extrapolation of results to management areas with similar characteristics, thereby avoiding unnecessary analysis repetition. On the other hand, site-specificity is essential in the operational level planning in Level III and the planning for individual events in Level IV. Higher resolution analysis is restricted to the fire program options which warrant that degree of detail. The cost would be prohibitive if the model and data resolution of Levels III or IV were used to evaluate the broad scope of program options which must be addressed in Level I.

The similarities ensure consistency and efficient progression of closer resolution analysis on a successively narrower range of program options. The differences permit the tailoring of the various models to the questions pertinent at the respective Steps of the integrated planning process.

Modifications

Although the parameters which affect analysis complexity are fairly obvious, such as high fire occurrence levels and high resource values, the optimum relationship between fire program complexity and analysis sophistication is not (*table 1*). The cost of added fire program information should be no greater than the opportunity cost of a less than optimal decision that would result from missing information. In the absence of a complete sensitivity analysis, proposals for complexity-related analysis modifications are judgmental.

Level I is needed in areas of moderate complexity, but Level II is not. The Level I first approximations are sufficient. Level III is needed in areas of moderate complexity, but relatively more reliance may be placed on historical experience. The range of fire program options addressed at all levels can be reduced. If the individual fire event is moderately complex, the first approximations derived from earlier analysis levels may be refined by judgment rather than by analysis.

Level I analysis is still important in low complexity areas, but the number of program options evaluated can be reduced even further. Level I can also be reduced from a full probability model to an estimate of expected values alone. The Level II analysis can be dropped, just as it was under moderate com-

Table 1—*Effects of fire management analysis complexity on management objectives, by levels of fire management analysis*

	Fire management analysis complexity		
Level	High	Moderate	Low
I	Not site-specific Numerous fire program options Full risk consideration	Same as for High with fewer options and data resolutions	Few fire program options Expected value output only Lower data resolution
II	Site-specific refinements of Level I results Special contraints included program options	Remove Level II analysis model Consider special constraints in Level I or in the interdisciplinary team	Same as for Moderate
III	Very site-specific estimate of cost, location, and timing Moderate number of fire program options	Less site-specific Fewer fire program options	Derive output from judgmental prorating of historical information using Level I output
IV	Very site-specific and time-specific Use Level I and II output Moderate number of fire program options Full risk consideration	Expected value results only Use Level I and II output as input with less refinement	Few fire program options with onsite judgment adjustments

plexity and the Level III analysis model can be greatly simplified or perhaps removed. The mathematical analysis in Level III could be replaced by judgmental adjustment of Level I output and site-specific historical experience. In a low complexity fire event, Level IV would evaluate fewer options and greater reliance can be placed on analysis of hypothetical fire events.

CONCLUSIONS

The objective of a fire management analysis is to identify the program option that contributes most to the integrated management objectives. If the fire management analysis models are developed in the absence of a complete system design, major analysis gaps will occur in some places, and substantial overlaps will result elsewhere. The analysis and data resolution may be mismatched with the program options and the range of options tested may be too narrow.

This report describes the framework for a set of interrelated models, each one tailored to answer questions relevant at various steps of the resource management planning process. The analysis efficiencies that can be gained from the use of a completed set of models are likely to far outweigh their cost of construction.

The conceptual framework of the system described refines and extends previous work. Although currently available models approximate these design criteria for some of the components, the complete system is not yet operational. Full development of the framework will improve complementarity among analyses and reduce analysis overlap, ensuring that particular analyses are performed at the most efficient point in the sequence of questions answered during planning.

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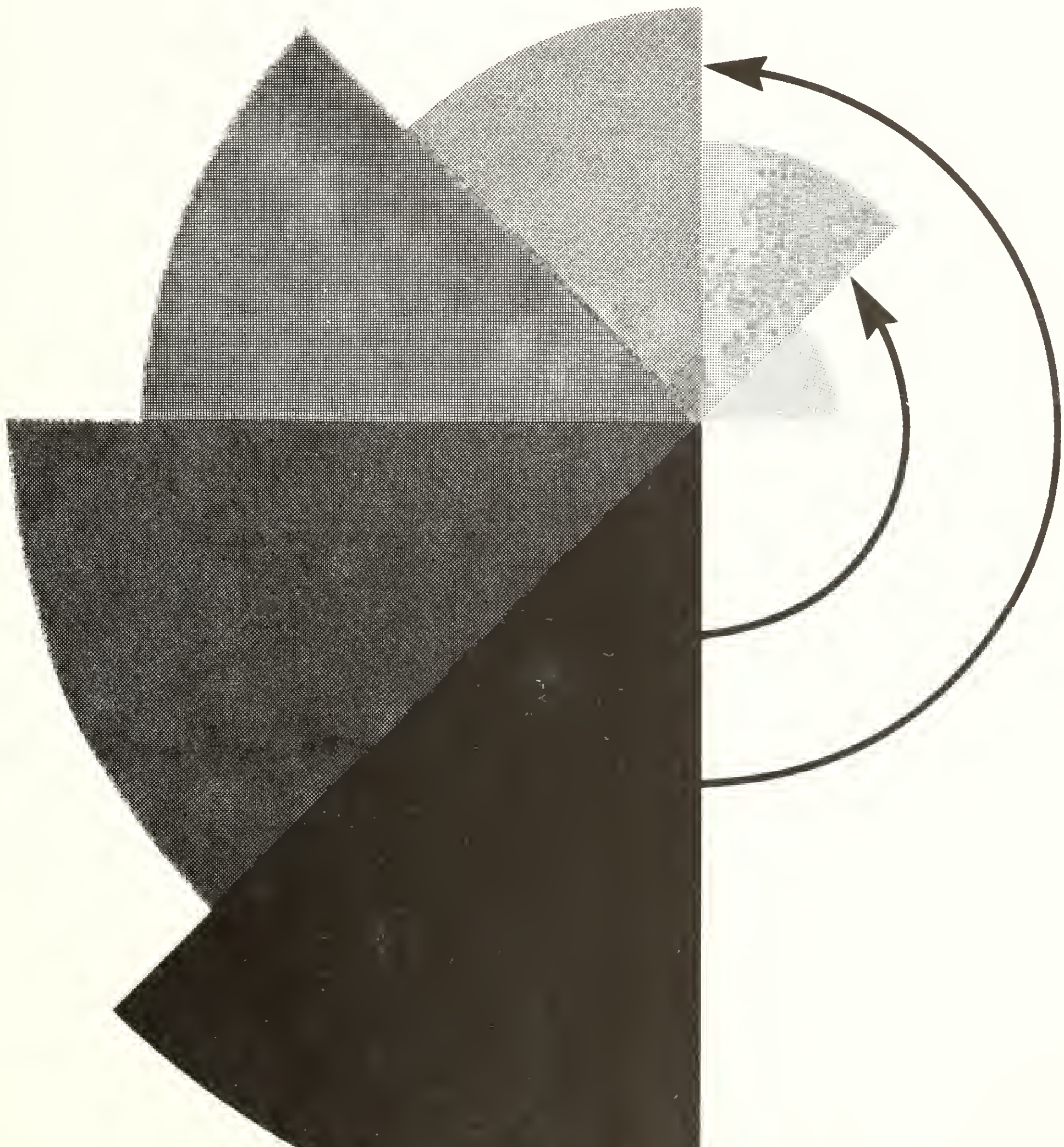
General Technical
Report PSW-75

Economic Efficiency in Forest Service Program Development

Robert J. Hrubes



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Publisher

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

January 1984

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CONTENTS

The National Forests are administered by nine regional offices of the Forest Service, U.S. Department of Agriculture. The process by which organizational capital—budgets, workforce—is focused and translated into management actions is governed by a complex and linked framework of planning systems. Activities that contribute to the system range from long-term goal identification, such as found in Resources Planning Act (RPA) planning, to setting up work plans for discrete, identifiable sets of actions. The annual program planning and budgeting (PP&B) process provides the link between long-term goals, objectives, and targets and specific work plans for the field units—the National Forests and their Ranger Districts.

Program planning and budgeting, an overlapping 2-year cycle initiated annually, consists of six major steps ranging from development of national direction to budget implementation and feedback evaluations (*fig. 1*). As the shortest-term planning process, PP&B functions to implement long-range plans such as forest land management and RPA plans. Annual expenditures on projects of work, as determined in the PP&B process, should—over the longer term such as a decade—lead to program output levels that meet commitments set forth in RPA and land-management plans (LMP). But the possibility always exists that commitments cannot be met because of limited budgets over an extended period. Consequently, a feedback linkage is needed to ensure that long-range targets are modified when necessitated by the budgetary record.

The focus of this report is the process and associated analyses that lead to the development and selection of the annual program at the regional level, because it is at the regional program formulation stage where tradeoffs among alternative program funding levels are initially addressed. Examination was focused on three representative Forest Service Regions: the Southwest (R-3), Pacific Southwest (R-5), and Southern (R-8). The study was designed to determine the extent to which formal consideration is given to arriving at an economically efficient program. Field interviews were concentrated on four program areas: nonsale reforestation, nonsale timber stand improvement, wildlife habitat improvement, and forest administration and other construction (FA&O). No attempt was made to document exhaustively the decision and analysis processes for all program areas that comprise regional-level program formulation.

Other studies pertaining to program development and budgeting have or are being conducted by the Forest Service. The most recent effort is the National Productivity Improvement Study of PP&B conducted by a Productivity Improvement Team (PIT) (U.S. Dep. Agric., Forest Serv. 1982a). Although similar in subject matter, the study reported here differs from the PIT study in a fundamental manner. The PIT study addresses efficiency in terms of the organizational costs associated with the process of arriving at annual program budgets. The study tried to identify means of assuring a more cost-efficient operation of the process, with efficiency defined as the least possible impact on organizational resources (for example, person-hours of effort and disruption of other activities). But the PIT study did not concern itself with the efficiency of the resource decisions and associated actions that are implicit in an annual program budget, the definition of efficiency that is of concern in this present study. While this study made no attempt to directly assess the efficiency of past resource decisions, it did attempt to determine the extent to which current analyses (if any), conducted as part of the PP&B process, contribute to efficient resource decisions.

METHODS

Decision Criteria

This study is concerned primarily with the efficiency—total benefits less total costs—of resource decisions. This orientation is in response to the objective voiced by R. Max Peterson, Chief of the Forest Service, of increasing the productivity of Forest Service programs. Beyond the Chief's current concern, direction to consider program costs and associated benefits is found in various legislation such as the RPA and the National Forest Management Act (NFMA). Section 1930.2 of the Forest Service Manual states that "The objective of program development and budgeting is to ensure . . . effective allocation of funds, targets and employment ceilings." What does "effective allocation" mean in this context? Coupled with the Chief's concern and associated direction, it clearly means that the allocations should maximize net benefits, subject to other considerations. But, efficiency is only one of several valid criteria that should play a role in annual program formulation. The other considerations that help shape, in varying degrees, program allocations respond to factors both internal and external to the Forest Service. They include:

Program balance—Within a program area, there exists a traditional concern over the equity of the geographical

disbursement of funds. It is generally held within the agency that each field unit deserves some minimum funding level. The distribution of program benefits among forest users and recipients is considered, also.

Temporal stability—In deference to workforce management and the welfare of employees who might be displaced, pronounced changes in program funding over a short time period are generally disfavored. Concern for the welfare of program beneficiaries outside the agency also contributes towards a tendency to favor no changes.

Political feasibility—Closely related to the motivations behind temporal stability are concerns for the public responses to Forest Service actions. Political feasibility considerations also favor avoiding change. Political feasibility should be defined broadly enough to include the reactions of Forest Service employees to changes in program allocations. Construction programs for FA&O are particularly sensitive to political considerations.

The extent to which these other considerations preclude or modify the attainment of efficient resource allocations in the sampled regions will be discussed later. On a region-by-region basis, it is a question of the relative weights attached to each decision criterion, both explicitly as formal decision elements and implicitly through the structures of the program development processes.

In each of the three study regions, the program analysis and formulation phases of PP&B were investigated through field interviews with Regional Office (RO) personnel and review of national and regional program planning documents. Interviews were conducted in September and October 1982. Two basic observations emerged from the interviews and associated documentation:

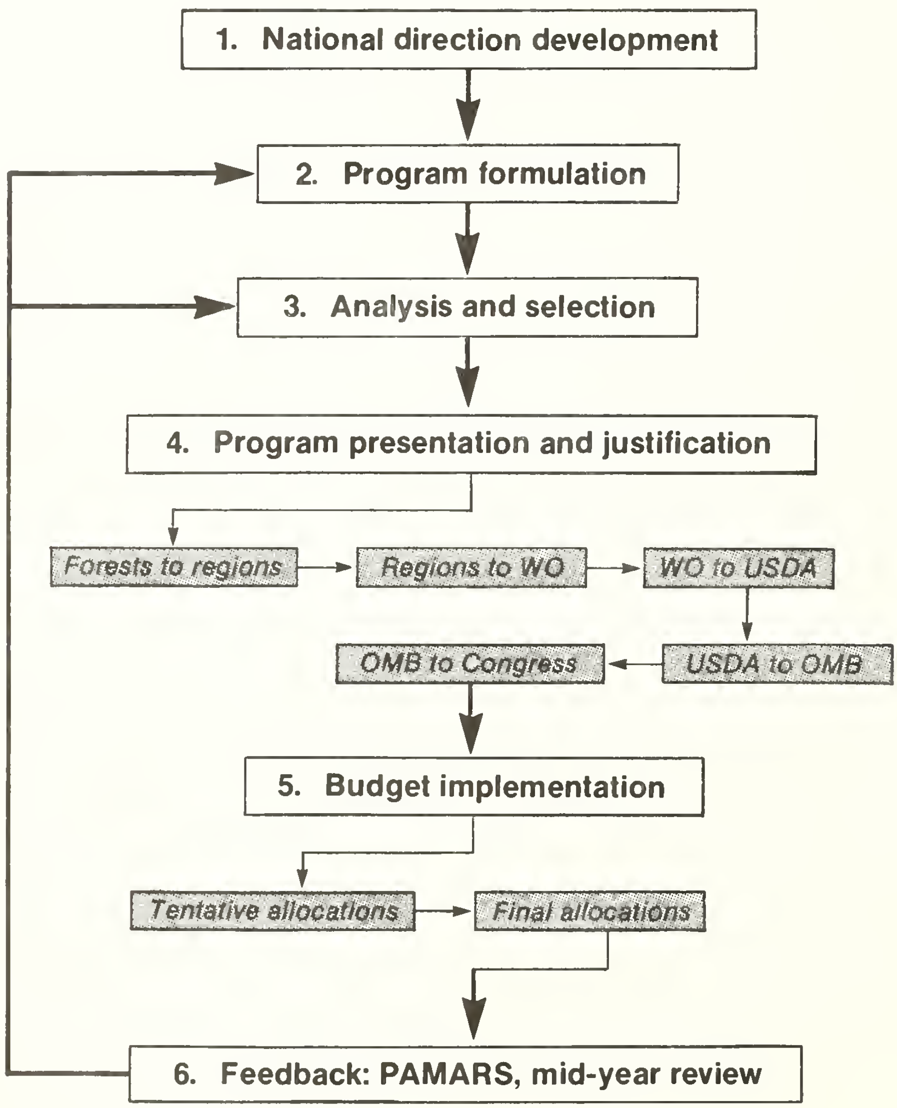

Figure 1—Program planning and budgeting include six major steps ranging from development of national direction to budget implementation and feedback evaluations.

• A wide variation exists across regions in the structure (procedural and organizational) for program formulation and in the type and extent of analysis supporting program formulation. Within a region, wide variation also exists across program areas in the type and extent of supporting analysis.

• The level of economic analysis is generally limited and insufficient to investigate adequately the relative economic efficiency of alternative programs.

Interviewees generally recognized the shortcomings of current analyses and a desire to upgrade the level of economic analysis that supports program formulation in their region.

To facilitate a discussion, the three regions' approaches to program formulation will be compared with an idealized approach that is not now used but, if used, would assure sufficient consideration of the relative economic efficiency of alternative programs. By presenting the discussion in this format, the variability in current approaches and their inadequacies can be more easily revealed.

A Model for Efficient Program Allocations

For reasons of clarity, the planning problem is rephrased from "program formulation" to "project selection." The task is to select a set of projects within a set of program areas (for example, timber, range, wildlife) that will comprise a regional annual program of work. Associated with a regional program, as defined by a set of projects, is an overall budget level and a set of targets and spending limits that are specific to program areas. Alternative regional programs, each with a different budget level, targets and spending limits, are elaborated. It is assumed that the candidate projects from which a program will be selected are themselves efficiently designed to accomplish the desired results at minimum cost. The process by which this is assured is "project evaluation" and falls outside the scope of this study.

At the regional level, targets and spending limits are set by the Washington Office (WO), but they may not be stipulated for all program areas. For purposes of this model, WO-derived targets and spending limits are treated as exogenously set parameters. Generally, however, targets and spending limits should reflect a negotiation between the regions and the WO that includes a consideration of their effects on efficiency. This is important because targets and spending limits reduce the discretion or latitude in the selection of projects that could comprise an efficient (net benefit maximizing) program.

The problem of project selection is formulated as a 0-1 integer program where the objective function involves the selection of projects from the set of all candidate projects that will maximize the difference between benefits and costs subject to targets (lower bounding constraints) and spending limits (upper bounding constraints). In this theoretical example are I subunits (forests), J program areas on each forest, and up to K candidate projects for each program area on each forest. Maximize:

$$\sum_{i=1}^{I} \sum_{j=1}^{J} \sum_{k=1}^{K} X_{ijk} (V_{ijk} Q_{ijk} - C_{ijk}) \tag{1}$$

subject to:

Program area-specific targets

$$\sum_{i=1}^{I} \sum_{k=1}^{K} X_{ijk} Q_{ijk} \geq T_j \tag{2}$$

Program area-specific spending limits

$$\sum_{i=1}^{I} \sum_{k=1}^{K} X_{ijk} C_{ijk} \leq B_j \tag{3}$$

Region-wide budget limit

$$\sum_{i=1}^{I} \sum_{j=1}^{J} \sum_{k=1}^{K} X_{ijk} C_{ijk} \leq B \tag{4}$$

All $X_{ijk} = 0,1$, a set of control variables restricted to 0-1 integer space, one variable for each candidate project in the region.

The 0-1 stipulation on the X_{ijk}'s requires that projects be fully funded (selected) or not at all. Each candidate project has associated with it an output level, Q_{ijk}, and a total cost, C_{ijk}. Associated with outputs are unit valuations (for example, prices or shadow prices), V_{ijk}, that should approximate the marginal social valuation of each output. An implicit assumption in this mathematical programing formulation is that the projects (columns of the matrix) are independent. In reality, complete independence of projects is not the situation. Some timber management, roading, and developed recreation projects, for instance, may be interdependent. Because of these violations, the IP solution is best viewed as an approximation of the optimal (that is, economically efficient) selection of projects.

Each integer program conforming to this general formulation can have a unique combination of values for B, B_j's, and T_j's. Therefore, it is a straightforward iterative process to generate alternative regional annual programs. (The term, program, in the context of mathematical programing should not be confused with its use in the context of program planning.) The currently favored approach to generate a set of incrementally augmented regional programs that build upon a base level is easily modeled through a succession of solutions to the integer program, beginning with the base (lowest) levels for B, B_j's, and T_j's and then increasing the levels of these parameters. To assure that projects are selected sequentially, the projects selected for a lower-level program, such as the base program, can be prewired into the solution for the higher-level (that is, greater funding and target levels) program by constraining the appropriate X_{ijk}'s to equal 1.

Data Requirements

To identify the set of projects that will comprise a regional annual program of work with this approach, data must be gathered on each candidate project for direct costs (necessary funding), outputs, and value of outputs (net of environmental costs). Because projects across all program areas are ultimately competing for the same budget dollars, their benefits must be directly comparable. Ideally, contribution to society's well-being (social utility) should be the object of measurement, as approximated by some combination of market artifacts (prices) and estimation based on professional expertise. It is not an easy task but one that is necessary for cross-project comparisons.

Additionally, information is needed to establish targets and spending limits. Because this task is accomplished by the WO, national-level priorities are given primary consideration. At the national level, an analysis must be made to arrive at a preferred and feasible distribution of targets and dollars among the nine Forest Service regions. This process, of equal significance to its regional-level counterpart, was not examined in this study.

Solution Characteristics

With this problem formulation, a solution algorithm is free to select projects that meet the targets and spending limits without regard to their distribution among the forests in the region. All projects within a program area regardless of location can compete against each other on their merits (net benefits) in the selection process. The order or priority by which projects are selected is determined by their contribution to the net benefit of the entire regional program. Projects with the greatest net benefit are selected first. Accordingly, the last project selected within a program area on a forest would not have a net benefit contribution lower than a nonselected project from that forest or any other forest. The net benefit contribution of the last project selected on each forest, therefore, would tend to be approximately equal, within a program area. If spending limits are not binding (that is, do not influence the solution), the equality of the marginal (last selected) project extends across program areas. This result is equivalent to the equimarginal principle (first-order conditions) of economic optimization found in a continuous, as against integer, problem formulation. The larger the number, and the smaller the size of candidate projects, the more closely will the solution fit these characteristics. With large projects, lower priority but smaller projects may be selected to conform to spending limits.

The projects selected under this formulation would maximize the economically definable net benefits to society of budget expenditures for resource programs in the region being analyzed. But criteria other than economic efficiency are not considered. For this reason, there is no assurance that such a regional annual program would be desirable or operationally feasible. It is likely that some forests would have no projects selected for funding within a program area. As will be discussed, criteria such as equity and stability, that are manifested in distributional specifications on project selection, are an important component of program formulation in the sampled regions.

Beyond documenting these nonefficiency considerations and demonstrating how they fit into or modify the theoretical model, an important question broached in this paper—but one that must be answered by agency policymakers—is this: *To what extent should efficiency be sacrificed in order to respond to other considerations?*

RESULTS

In general, the program planning processes of the three regions do not assure that projects are selected with regard to economic efficiency. If the net benefit maximizing formulation were used in these regions, the resulting programs would probably look much different from those currently generated. That is, the regions are currently accepting a cost in terms of foregone net benefits. Two basic reasons for departure from the efficient solution are (1) other, nonefficiency considerations impose requirements on project selection that limit the feasible range of net benefit maximization, and (2) the regions are not fully or properly using the discretion that remains for pursuing economically efficient programs.

Limits on Project Selection

Programing considerations that partially determine the annual selection of projects are generated both internally and externally to the agency. Appropriately, the program planning process at the regional level should respond to regional and local considerations that cannot be reflected in WO-generated targets and spending limits. Washington Office directives are defined in terms of program area totals without concern for the breakdown within a program area. The result is that projects are selected for inclusion in the annual program for a variety of reasons.

At lower levels of program funding, concern for a geographically balanced program has the single greatest effect on project selection. In all three study regions, every effort is made to select those projects the forest staffs submit as their base (minimum)-level program. Regional personnel think it organizationally and politically undesirable to generate a regional program in which some forests are not receiving a minimally sufficient level of funding (and project selection) in each program area. Other arguments for at least minimum-level programs on each forest are that they are necessary to (a) assure that multiyear programs such as plantations needing timber stand improvement are not aborted, (b) meet legal requirements, and (c) provide necessary support to other activities such as land manage-

ment planning. The projects making up a minimum-level program, therefore, are prewired into the selection process and are generally not required to withstand tests such as comparing their net benefits with those of other candidate projects. Ideally, the forests will submit minimum-level projects that compare favorably with their other projects, but it is not a requirement and not always the situation. By assuring that minimum-level projects submitted by the forests will be selected, the regions are also allowing for local-level priorities and issues to be considered in the programing process. Forests can be fairly assured that their highest priority projects will be funded if they are submitted as part of the minimum level. But, this fact further diminishes the likelihood that minimum-level projects will be the most efficient.

The desire for a geographically balanced program is, in part, a reflection of concerns over disruption of workforces and the costs of relocating employees to accommodate changing funding levels. These concerns, combined with an organizational tendency to do business in a way that "doesn't make waves," result in a tendency to favor program allocations that are not a significant departure from the previous year. To the extent that a geographically unbalanced program may be the most economically efficient, regional programs that reflect balance and minimal change from previous years result in opportunity costs of foregone net benefits. Examples of where it may be more efficient to simply ignore some forests in the selection of projects within a program area can be found in all three study regions. In Region 8, it is widely recognized that the coastal plain forests have a clear competitive advantage over forests in the Appalachian region in the production of timber. A similar situation exists with the forests of northern versus central California. To maintain, in the name of balance, at least a minimum-level timber management program on some Appalachian and Sierra Nevada forests is, perhaps, economically inefficient.

To reflect program balance requirements, the mathematical model developed previously must be modified by the specification of a set of up to I × J additional constraints that stipulate a lower bound on activity on each forest, by program area:

$$\sum_{k=1}^{K} X_{ijk}\, Q_{ijk} \geq M_{ij} \tag{5}$$

(M_{ij} = minimum output of program area j on forest i). By adding these constraints, the feasible range of program selection is reduced and the objective function (net benefits) may be compromised. But, by doing so, it is more likely that the resulting project selection will be operationally feasible. Let us add constraint set 5 to the idealized program selection formulation (equations 1-4), recognizing that unconstrained economic efficiency is not likely to result in socially desirable solutions.

At the regional level, the reduction in the discretionary level of project selection is not always in response to explicit and voluntary regional considerations such as maintaining minimum-level programs on each forest. In discussions with regional personnel, for instance, it became apparent that targets passed down from the WO often have the effect of removing from the regions any real opportunity to design programs with regard to efficiency, or any other criterion. When targets are set at prohibitively high levels, the dominant problem becomes one of finding enough projects just to meet the targets. The notion of picking the "best" projects with regard to their efficiency characteristics is essentially irrelevant; any and all available projects that can contribute to meeting the target are needed. In Region 5, for example, the reforestation target passed down from the WO for a recent year was increased by 3000 acres (1215 ha) per year, reflecting Congressional interest in quickly eliminating the nation's reforestation backlog. In recent years, the reforestation rate in this region has been about 12,000 to 15,000 acres (4860 to 6070 ha) per year. Regional timber personnel are scrambling to find enough land available (as judged by forest-level personnel) to meet the increased target level. Efficiency ceases to be a relevant issue in such a situation.

A more general statement of the high target situation is that in some program areas there are limited investment opportunities. Implicit in the concept of economically efficient program selection is the assumption that an excess of investment opportunities (projects) and a constraining spending limit prevent some candidate projects from being funded. The regional experience does not always conform to this structure: sometimes the availability of projects is more limiting than the availability of funds.

Conversely, spending limits can be so constraining that the selection of projects is limited regardless of the number of candidate projects. This is especially true when coupled with minimum-level funding requirements for each forest in a region. When WO-generated spending limits for a program area are very low and the regions seek to continue geographic balance by assuring minimum-level funding, all available funds are used up at the minimum-level, where no efficiency analysis takes place. There is then no opportunity to exercise much discretion in project selection. The wildlife programs in Regions 3 and 5 are current examples of this situation.

In situations of high targets or low funding, or when both exist simultaneously, regional program planners are faced with constraints that severely limit the feasible range of project selection. In fact, the constraints themselves can dictate the solution, effectively removing any discretion in project selection.

Other, externally generated, limiting factors that predetermine, to varying extents, the selection of projects include "earmarked" funds and coordination with plans and programs external to the Forest Service. With timber and wildlife programs, some revenues, by law (for example, Knudsen-Vandenberg funds), must be reinvested on the same geographic area from which they were generated. Similar stipulations hold for a portion of range betterment

monies as administered under the Federal Lands Policy and Management Act of 1976. Obviously, if these types of funds are to be spent, projects designed for the pertinent land areas cannot be tested for their relative economic efficiency against alternative projects on other land areas. Again, the feasible region of project selection is reduced.

The regions' latitude in project selection is also reduced by necessary coordination with other public land management agencies. Resource management problems and issues often cross property lines and require coordinated actions from several agencies. An example is the management of wildlife. Here, the Forest Service is responsible for habitat management while State agencies and the Fish and Wildlife Service, U.S. Department of the Interior, are primarily responsible for the management of the wildlife itself. These other public agencies have developed wildlife management plans (for example, State comprehensive wildlife and fish plans, recovery plans for Federal and State listed species). And for these plans to be fully implemented the Forest Service may be obliged to program supportive actions on pertinent National Forest lands. The result is that the allocation of another block of the annual budget is, in part, predetermined.

The funding (selection) of some projects is influenced by the availability of matching funds supplied by other agencies but available for use on the National Forests if accompanied by Forest Service expenditures. To use these matching funds for which restrictions—such as geographic-use limits, time limits—are included, some projects will require selection over others. The feasible range of project selection, again, is reduced. An example of the matching funds situation is the $1 million of, mostly matching, non-Forest Service funds used annually for fish and wildlife management in Region 5. Most of the funds are spent in cooperation with the California Department of Fish and Game, which has a strong voice as to where the monies are spent.

As indicated by these examples, one of the basic problems of regional program formulation is the high degree to which decisions on project selection are predetermined. Contributing factors range from external matching funds to the internal Forest Service policy of assuring a minimum-level funding for each National Forest. In terms of the descriptive mathematical model, these factors are all represented as additional constraints on the feasible range of project selection. What remains after their specification represents the opportunity to exercise discretionary project selection.

Use of Discretionary Opportunity

Opportunities to incorporate economic efficiency are further lost by the standard practice of before-the-fact disaggregation of the regions' targets and spending limits among the forests. The regions break down their targets and spending limits by specifying and then including forest-level targets and spending limits in the annual program formulation instructions sent to the forests. If a region's reforestation target sent down from the WO is 10,000 acres (4047 ha) for a particular tentative funding level, for example, the region disaggregates the 10,000 acres into targets sent down to the forests. The region's allotment of appropriated funds for that reforestation level is similarly disaggregated. Forest-level targets and spending limits are depicted in the mathematical model as:

Target for j^{th} program area on the i^{th} forest

$$\sum_{k=1}^{K} X_{ijk} Q_{ijk} \geq T_{ij} \tag{6}$$

Spending limit for j^{th} program area on the i^{th} forest

$$\sum_{k=1}^{K} X_{ijk} C_{ijk} \leq B_{ij} \tag{7}$$

Total spending limit for i^{th} forest

$$\sum_{j=1}^{J} \sum_{k=1}^{K} X_{ijk} C_{ijk} \leq B_i \tag{8}$$

The logic behind this approach has, as its basis, a desire to ensure, in advance, that the forests will submit program proposals that, in total, enable the regions to meet their targets, within the spending limits. The regions set forest-level targets and spending limits such that:

$$\sum_{i=1}^{I} T_{ij} = T_j \tag{9}$$

$$\sum_{i=1}^{I} B_{ij} = B_j \tag{10}$$

in which:

$$\sum_{j=1}^{J} B_j = B \tag{11}$$

Potentially, however, an unnecessarily high price has to be paid when regions use this approach "to cover themselves," so to speak. By casting out forest-level targets, the regions are implicitly making resource allocation decisions without benefit of supporting analysis. Each forest's relative efficiency in producing the various regionally targeted outputs needs to be compared with that of each other forest. It appears that the overriding influence on the breakdown of regional targets into forest targets is the breakdown in the previous year. The targets sent to the forests generally do not change in terms of the relative positioning of each forest in the region (that is, for a program area, forests with high targets generally remain so, relative to the forests with low targets). Although the initial establishment of the relative forest target levels, for some undetermined prior year, may have reflected relative forest productivity, there is no longer any such explicit consideration. The result is that the targets sent down to the forests may lead to a very inefficient distribution of the forests'

contributions in meeting the region's total targeted output for any given program area.

From an efficiency standpoint, it is better not to make before-the-fact resource allocations in the form of forest targets and spending limits. By imposing additional constraints on the feasible range of project selection, the total net benefits of an annual program are reduced. And from the standpoint of ensuring that regional targets and spending limits are met, these constraints are unnecessary. It is much better to distribute the forests' contributions to meet regional output targets according to the forests' relative efficiency of output production. Relative efficiency is defined by the cost-benefit characteristics of candidate projects (subject to lower bound limits reflecting consideration of operational feasibility). Such a distribution is approximated with the integer programing formulation defined by equations 1 through 5. The solution algorithm selects which forests should supply how much in meeting each of the region's output targets within the regional spending limits and program targets. And it is done in a manner that results in maximum net benefits.

Focusing now on the discretionary project selection that does take place, with special attention to the roles played by economic analysis and economic efficiency, the review of the study regions revealed a wide variation in the extent to which economics is used in the annual process of project selection. Variation is found between regions and across functional program areas within a region. No one region is clearly more advanced in the use of economics than the others—each region has its stronger points and weaker points.

A sample of situations at both ends of the spectrum—when economic analysis and efficiency play a dominant role and when they play no role—can effectively describe the variation. Additional sample situations will be discussed where economic analysis is recognized and attempted, but in a fundamentally flawed manner. Because this study was not designed to assess individual performances, the location and personnel of each sample situation discussed are not included.

In two of the three study regions, the annual selection of FA&O construction projects is subject to the most structured decisionmaking with explicit and formal consideration of project benefits and costs. A significant variation exists, however, in the extent to which benefits and costs are considered. And in the third study region, although the significance of relative costs and benefits is generally recognized, there is no formal elaboration of project net benefits or any other attribute. But, in general, each region engages in some form of analysis—either formal or informal—in each of the following areas:

Inventorying and updating of candidate projects:

- Candidate projects are developed and submitted to the RO in an organized and consistent manner.
- Regional personnel have a good understanding of what projects are available for funding.

Evaluation of project benefits and costs:

- The RO has information on each project's economic and other pertinent characteristics.
- Regional personnel give greatest attention to those projects that will generate the highest net benefits.
- Evaluations and analyses range from informal "show-me trips" in one region to formal elaboration of evaluation matrices in another region, known in the Forest Service as Kepner-Tregoe matrices (Kepner and Tregoe 1965).
- Projects are given priority and those with highest priority are funded, down to the spending limit.
- Priorities reflect cost-efficiency and other criteria.
- Analyses, prioritizing, and funding decisions are products of a team approach.

The content and quality of supporting analyses vary widely. The region that relies on informal understanding of the attributes of each candidate project (often by only one member of the RO team) is hardly comparable to those regions that develop formal and quantitative information on each project. But the consistent trait is that in each region the FA&O selection process is at least as competent as the selection process for any other program area within the region. The region with the weakest FA&O selection method generally is also weakest in the other program areas.

Another notable example of a project selection process that has incorporated cost-benefit considerations in a well developed, formalized method is found in range management in one of the three study regions. The procedures adopted avoid some of the shortcomings found in other regions and program areas. All candidate projects, even those of the minimum-level, for instance, are subject to cost-benefit evaluation. Only projects with positive net benefits are eligible for funding. But perhaps the most valuable attribute of their range allotment project selection procedure is the degree to which it is formalized and standardized. As such, it requires a consistent level of analysis of every candidate project. Further, it is a well documented procedure that allows easy tracking of the process and rationale leading to project selection.

The range allotment selection procedure evaluates project effectiveness in terms of three criteria: economic efficiency, environmental quality, and social impacts. Both market and nonmarket values are incorporated into a calculus that results in numerical indices by which projects are ranked. Benefits, costs, and associated project attributes—such as environmental quality benefit rating—are computed and recorded on standardized forms. The entire procedure has been incorporated as a range allotment project effectiveness handbook (U.S. Dep. Agric., Forest Serv. 1982b). In large part, this handbook could be generalized to apply to all program areas.

Conversely, the annual process of program formulation (project selection) in some program areas provides little or no recognition of the efficiency aspects of program funding, as defined by measures such as net benefits. In these

instances, the procedures by which funds are programed among the forests in a region are also usually ill-defined and informal. The dominant criterion governing program formulation seems to be a combination of giving to the forests what they request and essentially continuing the funding patterns of the previous year. When changes from the previous year are required, the general tendency is to increase or decrease the funding to each forest by the same percentage—an arbitrary, simple, and amicable approach that ignores the relative efficiency of programs in some forests over the same programs in other forests.

The direction to the forests generally is also sketchy, resulting in inconsistent project submittals. Some forests, by putting together better packages for submission to the region, have a better chance for funding. Economic attributes, such as unit costs, do not play a part in the process. Although this type of approach to project selection probably, over time, recognizes gross situations of comparative advantage for some forests over others in producing the desired outputs for the program area in question, the chances for significant inefficiencies in programing are high.

Most program areas examined in the three study regions fall in a middle ground where attempts are made to incorporate some elements of economic logic into project selection, but often in a flawed manner. Two of the most common areas where improper procedures are followed are (1) identification and measurement of program outputs and benefits, and (2) marginal adjustments in funding levels.

The first common problem area concerns the definition of unit costs. Two of the three study regions gather information on project unit costs and use these indices in project selection. But the manner in which unit costs are defined in some program areas fail to identify the relevant output from which costs per unit should be constructed. The best examples are in the timber management program: reforestation and timber stand improvement (TSI). The relevant average cost figure for comparing the efficiency of alternative projects is the cost per unit of output generated. In both reforestation and TSI, the appropriate output is the volume of marketable timber ultimately produced as a result of actions taken (investments) in those areas. But in the study regions, unit costs are defined in terms of the cost per acre treated, either with TSI or reforestation. Although it may be tempting to view an acre treated as the output and, hence, benefit of a TSI or reforestation investment, it is not the product that translates directly to the benefit to consumers and to society. The benefit to society is not a "treated acre" but the usable products generated by treating that acre. With timber management investments, the relevant output is the timber volume generated. To ignore the benefit side of the ledger can lead to inappropriate investment decisions (that is, inappropriate project selections). The costs per acre treated of a particular TSI project may be low relative to other projects, but if that project is not likely to result in the greatest increases in marketable timber, other projects may be a better investment.

Perhaps part of the current resistance to defining unit costs in the appropriate manner results from the difficulties in estimating the input-output relationships that translate acres treated to timber volume generated. These production relationships are complex because of variables such as the long time periods involved that result in significant degrees of uncertainty. If the usable product of treated acres is defined as the marketable volume of timber that actually can be offered for sale, the production relationships are further complicated by the complexities of the allowable harvest scheduling procedures. But these problems should not be considered insurmountable; to cut off the analyses at the input stage (acres treated) because of estimation difficulties should be considered unacceptable.

Another common problem area is the treatment of funding levels at the margin. Two examples of this situation were observed in the study regions. A common practice is to prorate program funding level changes, within a program area, by a constant percentage across all forests in a region. Suppose WO direction states that the regional program funding for level 1, for instance, is to be 15 percent less than the previous year. Commonly, regions elaborate that program by reducing the spending limit (funding level) by 15 percent on each forest. It is a procedure easily administered at the region and considered equitable by the forests. But from the standpoint of an economically efficient program, it is better to meet the 15 percent overall funding reduction by cutting funding on the least productive (least efficient) forests, thereby protecting the budgets of the forests with comparatively efficient projects. The object is to adjust the forests' funding levels so that marginal net benefits are approximately equal, across forests, where marginal net benefits are defined by the last project funded.

The failure to conform to the equimarginal net benefit principle was also observed in one region's timber management program. When selecting projects to comprise a given regional program level (for example, level 0, 1, 2, . . .), the highest priority project from each forest is selected before lower priority projects from any forest are considered. This procedure of selecting all projects of the same priority level before considering lower priority projects is followed until the overall program funding level is reached. From an efficiency standpoint, two problems result from this approach: the forests' rankings of their candidate projects may not be coincident with a ranking on the basis of net benefits; the regions' practice of funding all priority 1 projects before any priority 2 projects, and so on, ignores the strong possibility that a lower priority project on one forest may have greater net benefits than higher priority projects on other forests. Even if it is decided to accept each forest's priority ranking, it should be possible to select, for example, priority project 1 through 4 on one forest and only the first priority project on another forest.

DISCUSSION

This report has addressed the ways that current program planning procedures diverge from an idealized approach in which economic efficiency is the dominant concern. But economic efficiency is not the only consideration that can or should validly influence annual project selection. The decision as to the priority that should be attached to various considerations—for example, efficiency, equity, stability, political feasibility—is ultimately the responsibility of Forest Service policymakers. But it seems apparent, on the basis of the findings of this study and the stated concerns of the Forest Service Chief, that regions must begin to place more emphasis on the economic efficiency of annual programs. And this increased emphasis must be associated with valid procedures.

The level of importance that should be attached to economic efficiency cannot be determined in this paper. It is appropriate, however, to suggest as a minimum that regions can and should estimate the costs of responding to other considerations in terms of net benefits foregone by not giving exclusive consideration to economic efficiency. Such costs can be estimated by initially formulating programs with exclusive regard to economic efficiency, as outlined by the mathematical model in this paper. These "economically efficient programs," which serve as a starting point, can then be modified to include other criteria that are judged to merit consideration in program planning. By initially developing this "unconstrained" program as part of a sequential program planning process, it is possible to easily identify the extent to which efficiency is sacrificed (as measured by reductions in net benefits) because of other considerations. In the context of the 0-1 integer programing formulation that has been outlined here, this is done by resolving the solution algorithm with the specification of additional, nonefficiency induced constraints. The extent to which nonefficiency induced constraints should be part of the formulation must be determined by Forest Service policymakers.

Aside from the issue of generally upgrading economic efficiency as a decision criterion, attention must focus on reducing the variability in the quality of economic analysis across regions and program areas. A primary reason for greater consistency in the quality of analyis is that, ideally, the same types of analyses should be conducted at the WO level; the WO-to-regions disaggregation is analogous to the region-to-forests disaggregation. For regions to be fairly compared in the process of establishing regional program spending levels and targets, the WO needs to have consistent economic data from each region.

The current variation in the extent and quality of economic analysis in the field is evidence that WO direction needs to be modified. A review of program budget development instructions for Fiscal Year (FY) 1984 and FY 1985 reveals that instructions on economic analysis are sketchy, at best, with little continuity between years. The economic analysis instructions appear to be written by noneconomists; general concepts are crudely and, occasionally, inaccurately discussed with no accompanying direction to the field as to how the analyses are to be conducted. In contrast to the concerns voiced by the Chief, the FY 1985 instructions contain less detail and a generally reduced sense of importance towards economic analysis than do the FY 1984 instructions.

In talking with regional program planning personnel, the presence and influence of WO economic analysis direction was barely noticeable. Section 8.6 of the FY 1984 instructions states: "Subunits must build program submittals based on economic efficiency and effectiveness." As has been discussed in this paper, considerable improvement is needed in the regions' conformance to this directive. The direction and guidance from the WO must have stronger control and enforcement mechanisms to assure conformance. And perhaps even more important, the WO must provide more assistance to the field in the form of technical guidance so that the analyses can be effectively carried out. The deficiency is not found in the regions' acceptance of the policy of economic analysis as a crucial element in resource decisionmaking but in the ability of the regions to implement the policy.

Specifically, the WO needs to provide guidance on technical issues essential to annual project selection, such as how to carry out various economic analyses and how to set up a program planning framework that responds to economic efficiency. Well developed and established guidelines would also lead to greater consistency across program areas and regions. Currently, all regions have a regional economist position, but not in all regions is the position assigned to the program planning and budgeting staffs. And even in the regions where it is, the staff economists are working almost exclusively on land mangement planning (forest plans) and RPA planning with little or no time devoted to annual program planning. As the workload associated with the current wave of forest plans subsides, the regional economists may have more time to devote to annual program planning.

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The Forest Service, U.S. Department of Agriculture, is responsible for Federal leadership in forestry. It carries out this role through four main activities:

- Protection and management of resources on 191 million acres of National Forest System lands.
- Cooperation with State and local governments, forest industries, and private landowners to help protect and manage non-Federal forest and associated range and watershed lands.
- Participation with other agencies in human resource and community assistance programs to improve living conditions in rural areas.
- Research on all aspects of forestry, rangeland management, and forest resources utilization.

The Pacific Southwest Forest and Range Experiment Station

- Represents the research branch of the Forest Service in California, Hawaii, and the western Pacific.

Hrubes, Robert J. **Economic efficiency in Forest Service program development.** Gen. Tech. Rep. PSW-75. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1984. 9 p.

This report analyzes the procedures used in three regions of the Forest Service, U.S. Department of Agriculture, for selecting the projects that constitute their annual program budget. Personnel at the Southwest (R-3), Pacific Southwest (R-5), and Southern (R-8) Regions were interviewed during September and October 1982. Of special concern was the extent to which analytical methods are used that include an explicit and formal consideration of the relative costs and benefits of alternative program allocations. Current program development and budget processes were found to allow only limited discretion for efficient allocation of funds in many program areas. Contributing factors include both internal policy and practices that unnecessarily limit programing discretion and external commitments beyond the ready control of the agency. Where discretion does exist to develop annual regional programs with greater regard to economic efficiency, regions and program areas varied widely in the extent to which the opportunities are exercised. Deficiencies in internal policy direction, in the availability and consistency of project data, and in composition of planning staff are seen as contributing factors. A conceptual framework is developed that is designed to elaborate regional program alternatives that maximize the net benefits of program budget expenditures subject to regional targets and spending limits.

Retrieval Terms: program planning, economic efficiency, project selection

United States
Department of
Agriculture

Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-76

A System of Vegetation Classification Applied to Hawaii

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Acknowledgments:

The System of Vegetation Classification described in this report developed through the efforts of many persons. We thank all of the participants in the vegetation classification workshop held in Hawaii, 1982, especially botanists Gar Clarke and Linda Cuddihy of the Hawaii Division of Forestry and Wildlife. We also thank the manuscript reviewers, especially James D. Jacobi of the Fish and Wildlife Service, U.S. Department of the Interior. And finally, we thank Patrick G. Costales and Edwin Q. P. Petteys of the Hawaii Division of Forestry and Wildlife for their support in all phases of the work.

Publisher

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

February 1984

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CONTENTS

In Hawaii, the growing interdisciplinary responsibilities in resource management have resulted in a diversity of languages used to describe vegetation. Organizational and discipline oriented biases have given rise to unique systems for classifying vegetation—a reflection of the distinct information requirements of the various resource management and other technical groups. Through a variety of programs, the Hawaii Division of Forestry and Wildlife has management and protection responsibilities for more than half of the land area in the State *(fig. 1)*. There is a need for a classification system that names vegetative communities in terminology acceptable to the various disciplines it represents that can be used as a basic language among them.

This need was brought prominently into focus as the Division began work on a statewide multiresource forest inventory. The existing classification systems available in Hawaii were oriented either to a single function or extremely detailed with no hierarchy to a management level. Using a framework from the Vegetation Classification System for Southern California (Paysen and others 1980, 1982), criteria and nomenclature were adapted to develop the Hawaii Vegetation Classification System.

Representatives of these agencies participated in the development of the System: Hawaii Division of Forestry and Wildlife; Forest Service, U.S. Department of Agriculture; and Fish and Wildlife Service, U.S. Department of the Interior.

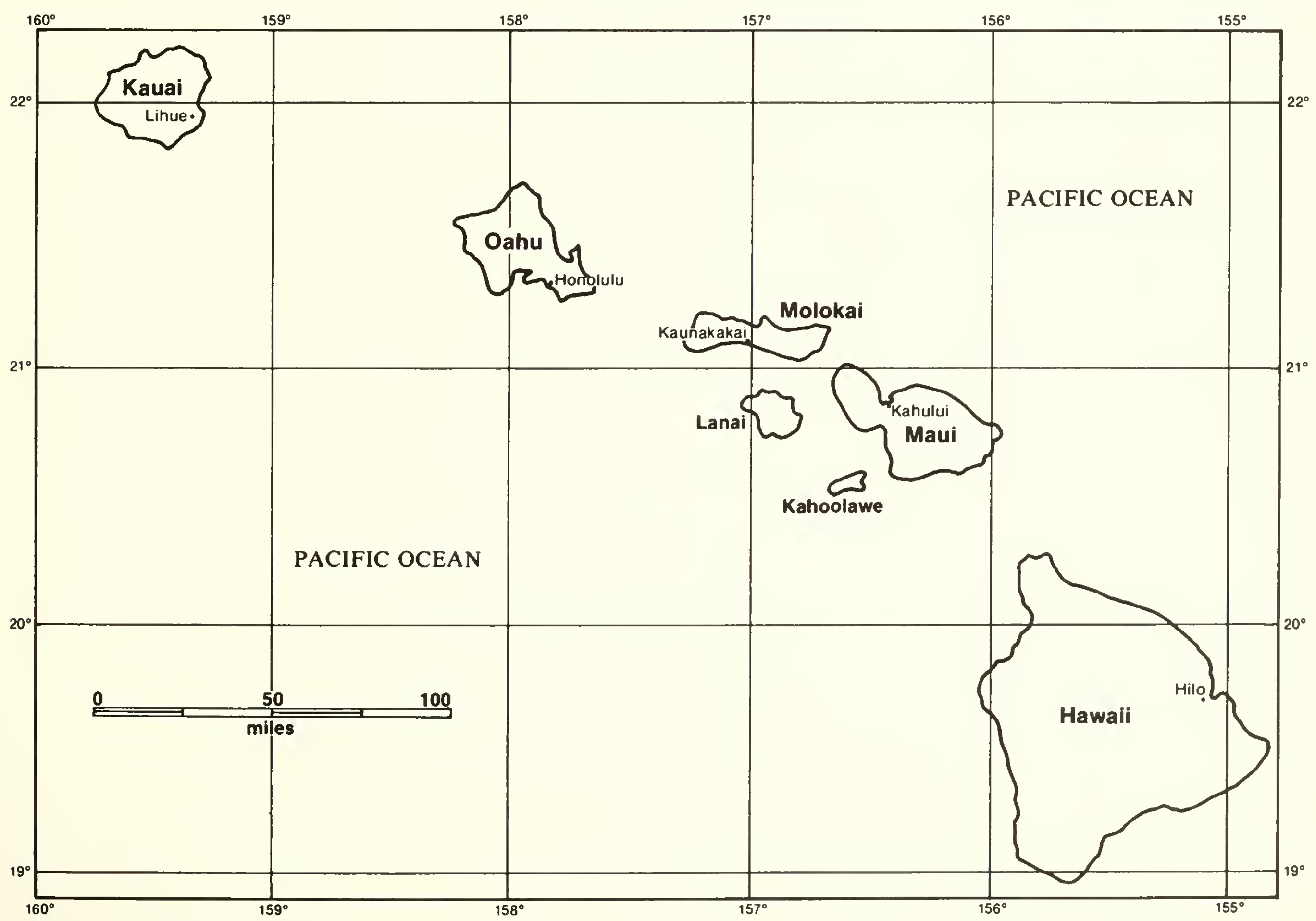

Figure 1—The State of Hawaii comprises 132 islands, reefs, and shoals stretching 2452 km southeast to northwest across the Tropic of Cancer. The eight main islands make up over 99 percent of the total land area of 1,670,500 ha.

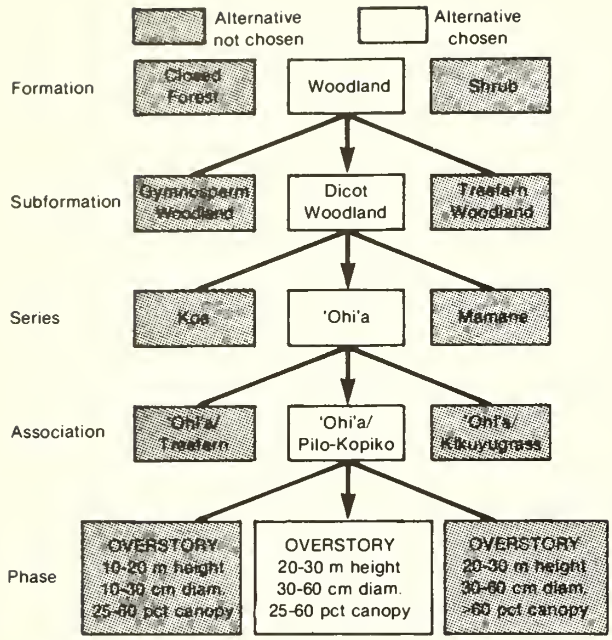

Figure 2—The hierarchy of the Hawaii vegetation classification system is shown here. The classification becomes progressively more precise as it moves through the five levels of the system.

The disciplines and resource management functions represented included research, inventory, wildlife, planning, management, and threatened and endangered species.

The System of Classification for use in Hawaii addresses a basic plant community unit with five levels of descriptive detail, the first four hierarchical *(fig. 2)*. The System is compatible with a proposed national land classification system (Driscoll and others 1982), and is also compatible with the International System of Vegetation Classification (UNESCO 1973).

The Hawaii System can be put to a variety of uses, but its chief function is to be a basic language. To fulfill this function, inherent in it are unique properties that make it flexible, functionally neutral, and applicable to both current and potential vegetation. The System provides a framework to identify missing gaps in our knowledge of the vegetative communities in Hawaii as well as a logical format to display and communicate this knowledge.

EARLIER CLASSIFICATIONS

Vegetation classification systems have been a necessary byproduct of inventory, mapping, land classification, or ecosystem classification. Vegetation classification schemes have been developed for use in local areas, such as the island of Oahu (Egler 1939), and for mapping vegetation on a global scale (Fosberg 1967, UNESCO 1973). Some systems were developed to provide a framework for describing the structure and dynamics of vegetation (Braun-Blanquet 1932), while others are used to place vegetation growing sites into categories defined by functional resource management criteria.

In Hawaii, although early explorers gave accounts of vegetation observed on the Islands, Hillebrand (1888) was the first to classify vegetation, recognizing five zones largely on the basis of elevation. A strand vegetation zone was added to this classification as well as moist windward and dry leeward subzones (Rock 1913).

In his ecological and floristic studies in Kipapa Gulch, Oahu, Hosaka (1937) recognized six plant zones on the basis of climate and flora. Elaborating on Hosaka's classification, Egler (1939) proposed for Oahu a series of vegetation zones that are correlated with differences in precipitation and insolation.

In their ecological work on the island of Hawaii, Robyns and Lamb (1939) used a vegetation classification based on assumed climax formations and physiognomic types of forest, parkland, shrub, moss-lichen, and bog. They also set up a new general classification of altitudinal zones for the island of Hawaii.

A zonal vegetation classification was applied to all the major islands on the basis of characteristics of the vegetation itself, as correlated with rainfall at the lower elevations and temperature at the higher elevations (Ripperton and Hosaka 1942).

Three recent vegetation mapping projects have added to classification efforts in Hawaii. Forest type maps of the State identified vegetation units by cover type, growth productivity, density, and stand size (Nelson 1967). A vegetation map of Hawaii Volcanoes National Park identified 31 major vegetation units by cover type, canopy height, and crown cover in six different macroclimates (Mueller-Dombois and Fosberg 1974). Statewide vegetation mapping of native Hawaiian vegetation separated vegetation units by tree crown cover and height, species composition and dominance, general habitat type, and understory species composition (Jacobi 1978).

CLASSIFICATION SYSTEM

The five levels of descriptive detail in the Hawaii System are Formation, Subformation, Series, Association, and Phase *(fig. 2)*. Of these, the first four levels are explicit members of a formal hierarchy.

Each level in the hierarchy is a more generalized version of the level below it and can also be characterized by criteria established for that level (floristic, physiognomic, or morphological).

The easiest way to understand the relationship between the various levels of the system is to start with its basic unit, the Association, and work up the hierarchy. In this aggregating system, the classes at a given level in the hierarchy are grouped according to prescribed similarities to form classes of the next higher hierarchical level.

Association

A plant Association is an assemblage of plants with a characteristic floristic composition that grows under a uniform set of environmental conditions. A plant Association is recognized by its several characteristic dominant species within defined vegetation layers.

The term ''Association'' has been used elsewhere to describe only those plant communities that are at climax or ultimate site potential. As Paysen and others (1982) have indicated, however, other authors have used Association to identify communities that are not necessarily at climax. Association in this report refers to plant communities that are relatively stable in terms of their persistence in a given vegetation system. A vegetation system comprises the plant communities and community dynamics that characterize a distinctive vegetation, and lend identity to an extensive ecosystem or floristic zone (Paysen and others 1982). The persistence of an Association may be attributed to its long-term occupancy on a given site or to its consistent occurrence over space or time in a given vegetation system. An Association, therefore, may be frequently observed on a landscape or may be a consistent invader after disturbances.

Associations have no community size (areal extent) limitations; yet areal limits in a given situation are imposed by the class boundary criteria for each formation, and by the cover dominance criteria applied to each layer when naming the Association.

In field testing the system, we found many Associations in Hawaii having distinct vertical layers. One plant, however, does not constitute a layer. A completely expressed layer, or a change of composition within a layer, must comprise more than a few random individuals on a landscape. Certain areas in the rain forest, for example, consisted of five layers: two overstory tree layers (5-30 m), a midstory treefern layer (3-5 m), an understory fern layer (0.5-2 m), and a ground cover layer *(fig. 3)*. Within each layer two or more species were present.

Adequate field study is essential for identification of Associations. Individual Associations may exist as a result of environmental interactions among stable and unstable biological and physical factors that include human influences. Two adjacent stands of 'ohi'a (*Metrosideros polymorpha*) on different volcanic substrates in the Hilo Forest Reserve, for example, may represent different Associations within the 'Ohi'a Series because of different species composition and dominance in the understory layers. Similarly, logging disturbance may have affected the vegetation composition of adjacent koa (*Acacia koa*) stands on the windward side of the island of Hawaii, and different Associations in the Koa Series might be identified.

Figure 3—An example of the vegetation layers found in many 'Ohi'a (*Metrosideros polymorpha*)/Treefern (*Cibotium* spp.) Associations.

Figure 4—Two commonly found Series of strand vegetation found in coastal areas. The foreground is the Ilima (*Sida* spp.) Series of the Forb Subformation, Herbaceous Formation. The Naupaka-kuahiwi (*Scaevola chamissoniana*) Series of the Shrub Formation is in the background.

Series

The Series can be thought of as generalizations of plant Associations. All plant Associations with given dominant overstory species constitute a Series that is named by that species *(fig. 4)*. This category does not imply greater heterogeneity than the Association. It still recognizes the basic plant community, but does so at a general level. It also allows

Figure 5—An 'Ohi'a/'Ama'uma'u Association of the 'Ohi'a Series found in Haleakala National Park on the island of Maui. This Association is important habitat for many of the rare Hawaii forest birds found in the park.

for reference to an entire set of floristically related communities. A particular plant community may be recognized as an 'Ohi'a/'Ama'uma'u (*Sadleria cyatheoides*) Association, for example, or it may be recognized as an element of the 'Ohi'a Series *(fig. 5)*.

In the use of this system, the identification of any hierarchical level should not be treated with a greater degree of precision than is warranted by current management needs or by requirements for advancing the state of our knowledge. A Series may be recognized, but may be sufficiently described for a given purpose by giving it a Formation level name. A Lantana (*Lantana camara*) Series, for example, may be described as an element of the Shrub Formation if further precision is not warranted *(fig. 6)*. Also, a generalized landscape description need not include the enumeration of all Associations that are found on it *(fig. 7)*.

Subformation

A Subformation is an aggregation of Series with a given physiognomic character and a particular stem and leaf morphology in the dominant overstory species. The following key may be used to identify the physiognomic and morphologic characteristics of the Subformations of the Hawaii Vegetation Classification System:

	Physiognomic or morphologic characteristic:	*Go to number*
1	Dominant plant species (overstory) woody	2
1	Dominant plant species herbaceous	10
2	Height of dominant species in overstory ≥ 5 m	3
2	Height of dominant species in overstory < 5 m	Shrub
3	Canopy cover of overstory > 60 percent	4
3	Canopy cover of overstory 25 to 60 percent	7
4	Dominant plant of overstory a treefern	Treefern forest
4	Dominant plant of overstory not a treefern	5
5	Dominant plant of overstory with seeds borne on upper side of open scales that are often produced in cones	Gymnosperm forest
5	Dominant plant of overstory with broad leaves bearing flowers	6
6	Dominant plant of overstory a monocot	Monocot forest
6	Dominant plant of overstory a dicot	Dicot forest
7	Dominant plant of overstory a treefern	Treefern woodland
7	Dominant plant of overstory not a treefern	8
8	Dominant plant of overstory with seeds borne on upper side of open scales that are often produced in cones	Gymnosperm woodland
8	Dominant plant of overstory with broad leaves, bearing flowers	9
9	Dominant plant of overstory a monocot	Monocot woodland
9	Dominant plant of overstory a dicot	Dicot woodland
10	Stand requires mechanical support of water to maintain physiognomic integrity	Aquatic
10	Stand does not require the mechanical support of water	11
11	Dominant species grass or grasslike	Graminoid
11	Dominant species not grass or grasslike	12
12	Dominant species cryptogamic (ferns, mosses, lichens)	Cryptogamic
12	Dominant species not cryptogamic	Forb

Figure 6—A valley bottom is choked with Lantana (*Lantana camara*), an aggressive exotic shrub introduced to the Hawaiian Islands as an ornamental.

Figure 7—A disturbed lower valley bottom on the island of Maui contains diverse populations of native and introduced plants.

Figure 8—A Hala (*Pandanus tectorius*) Series of the Monocot Subformation commonly found in coastal areas throughout the South Pacific. The leaves (lauhala), because of their toughness and pliability, are used by the natives of the Pacific for their homes, mats, baskets, and clothes.

Figure 9—An Uluhe or Staghorn Fern (*Dicranopter s linearis*) Series of the Cryptogamic Subformation. Uluhe grows in dense thickets at altitudes of 150 to 900 m. It smothers existing vegetation and prevents regeneration of other plants.

The Subformation category is included in the system to provide a set of classes that are more distinct than those at the Formation level; for example, treefern as opposed to monocot forests *(fig. 8)* or graminoid as opposed to cryptogamic *(fig. 9)*.

Formation

A Formation is an aggregation of Subformations with a given physiognomic character *(fig. 10)*. All Subformations characterized by an overstory of trees with a closed canopy (60-100 pct crown cover), for example, make up the Closed Forest Formation *(fig. 11)*. Four Formations are currently proposed for Hawaii.

Phase

Although Phase is treated as a category of the classification, it is primarily a vehicle to recognize unique demands made by specific disciplines, and for management application of the system. It is a means of addressing variability within a plant community and, therefore, only the general pattern of its use can be prescribed—strict definition of all potentially relevant Phases of plant communities is not practical in this report. The Phase level of description can be applied to any level in the hierarchy.

Phase descriptors are often chosen in relation to specific functional applications. A biologist interested in potential habitat for an endangered forest bird may define Phases of the 'Ohi'a/'Olapa (*Cheirodendron trigynum*) Association by the percent cover of nectar and fruit bearing tree species in the overstory layers. A forester who considers the noxious vine banana poka (*Passiflora mollissima*) in any Koa Series as an indicator of potential plant community disruption or developmental trend might use its presence as a phase descriptor

FORMATIONS	SUBFORMATIONS
Closed Forest	Dicot Forest Monocot Forest Treefern Forest Gymnosperm Forest
Woodland Forest	Dicot Woodland Monocot Woodland Treefern Woodland Gymnosperm Woodland
Shrub	Shrub
Herbaceous	Aquatic Graminoid Cryptogamic Forb

Figure 10—The 4 formations and 13 Subformations of the Hawaii Vegetation Classification System.

Figure 11—Two closed forest Formations commonly found in the coastal areas of Hawaii. A Coconut (*Cocos nucifera*) Series of the Monocot Subformation is shown on the right and the Kiawe (*Prosopis pallida*) Series of the Dicot Subformation in the upper left.

Figure 12—A Koa/Banana Poka Association found on the windward side of the island of Hawaii. Banana poka is an ornamental vine introduced to the islands in 1930 and has invaded more than 10,100 ha of native forest. It smothers the overstory canopy and successfully prevents regeneration of any other vegetation.

Figure 13—A Robusta Eucalyptus (*Eucalyptus robusta*)/Treefern Association on the island of Hawaii. More than 16,200 ha of eucalyptus plantations have been planted throughout the State.

(fig. 12). For a similar reason, signs of pig rooting and trails could provide the basis for a phase descriptor for the Koa Series.

The Phase category is also a logical means for relating developmental phases of different communities to one another across stages of community succession. A recently-burned koa woodland, for example, bears little resemblance to a woodland. The burned site is often dominated by elements of the Herbaceous Formation. But if koa seedlings and sprouts are present, the recent burn is a potential koa woodland in a pioneer stage (even though we classify it as a community in the Herbaceous Formation) and contains a particular Phase of a potential Koa Association. This knowledge allows us to describe the site in terms of its current and potential vegetation, and to incorporate short-term succession into our management planning.

Figure 14—A Bamboo (*Bambusa* spp.) Series found in Akaka Falls State Park on the island of Hawaii. Bamboo is used in Hawaii for crafts, Hawaiian dance implements, and fishing poles.

In using the Phase category to describe ecological stages, we are viewing the site in the context of what we know will probably happen to classify it more usefully for our needs. The system itself does not tell us anything about succession; it allows us to take ecological development and succession into consideration in applying the system for different purposes.

To describe Phase categories, a Phase code or index can be derived for specified purposes. Useful information that can be coded includes size, stature, and density information for desired vegetative layers. The use of a Phase code is not mandatory, but it can be useful for recording data.

NOMENCLATURE

The name given to an Association indicates the dominant and codominant species in each layer (*fig 13*). The combination of dominant and codominant species reflects the entire character of the Association—to the degree possible when only a few species names are used (*table 1*).

APPLICATION

Field classification of plant communities is the most obvious direct use of the system. The result of a field classification effort can serve as a basis for resource management planning, mapping, and environmental description. Many of the system's classification criteria correspond directly with the latest statewide native vegetation mapping project (Jacobi 1978).

The system can also serve as the common link between functional and technical classification systems, thereby facilitating communication between disciplines. An 'Ohi'a-Koa/Pilo (*Coprosma* spp.)-Kolea (*Myrsine lessertiana*)/Treefern/Lo'ulu Association should mean the same thing to every agency or group using the system. Results of classifying a plant community to one or all of the hierarchical levels of the system should be similar—within an acceptable degree of tolerance—no matter who is doing the classifying.

Although classification to the Series level and above is useful for many purposes (*fig. 14*), the system will become most meaningful to land managers when Associations are

Table 1—*Rules for naming plant Association and Series for the Hawaii Vegetation Classification System*

Situation	Nomenclature rule	Examples
	Associations	
Single-layered Association	Name by dominant species.	'Ohi'a; Koa
Multilayered Association	Dominant species in each layer will name the Association; start with overstory and end with herb layer, if one exists. Separate layer names with a slash (/). A maximum of five layers will be used.	'Ohi'a/Hapu'u pulu/Asplenium Association
Single species dominance in a given layer	Dominant species in the layer will supply the appropriate portion of the Association name.	'Ohi'a/Hapu'u pulu Association (overstory dominated by 'Ohi'a understory layer dominated by Hapu'u pulu)
Mixed species dominance in a layer (crown cover of each codominant species within 10 percent of other codominants)	Layer portion of Association name will be occupied by codominant species names—separated by hyphens. Where distinct synusia within a layer *characterize* an Association, they should be dealt with as codominants. A maximum of three codominants will be used for any layer.	'Ohi'a-Koa/Hapu'u pulu Association Pukiawe-A'alii Association
Sparse overstory layer—ecologically significant, but insufficient cover to define a Formation (10-25 pct cover)[1]	Include sparse layer parenthetically after the Association name.	Guava/Kikuyu Grass (Koa) Association
	Latin as against common name	
Local usage	Use common names if available for Associations and Series.	Guava/Kikuyu Grass (Koa) Association Kiawe Series
Official correspondence outside of administrative region, community documentation, scientific reports, or other	Use Latin or scientific names for Associations and Series.	*Psidium guajava/Pennisetum clandestinum* *(Acacia koa)* Association *Prosopis pallida* Series

[1]For overstory layers, these situations will carry over to the Series level. Series names should be developed accordingly; for example, Guava (Koa) Series.

identified. These can be identified only after plant distribution patterns and their associated environmental settings are carefully evaluated. Resource managers and researchers should work together in this process.

Direct application of the system carries no ecological connotations, nevertheless ecological science can be applied to identify and name plant communities.

The best way to develop an understanding of vegetation resource status and potential in Hawaii is to start with an accurate description of the vegetation that currently exists. This description should be at a level of precision that is sensitive to development trends and successional events. From this level, the interrelationships between vegetation and environmental controls can be evaluated for an understanding of vegetation dynamics as related to site development. This system of classification provides a means for developing such a description and for translating the picture of vegetation dynamics into resource management terms.

When Associations are identified, for example, their successional status should eventually be determined. Knowledge of the potential of a site has practical applications in local resource management and land-use decisionmaking. This system of classification provides a plant community framework that can be used to describe the successional pathway or set of alternative pathways that a site can follow. Such a description can help a manager to make reasonable decisions for allocating use of the site. An increased emphasis on managing koa for wildlife and wood products, for example, may bring about a response to increase production of koa in Hawaii. Knowledge of sites that have high potential for koa productivity will help to avoid fruitless attempts to force koa growth on marginal sites.

GLOSSARY

Broadleaf—Refers to all angiosperms with leaves that are *not* needlelike or scalelike. For this vegetation classification system, trees and shrubs that are *not* gymnosperms will be considered broad-leaved.

Canopy—The aggregate of tree and shrub crowns that provide a layer of cover; most often used in reference to tree crowns that provide an "overhead" canopy.

Closed forest—Generally, a unit of vegetation with overstory trees whose crowns are mostly touching. Plant communities with trees having a crown cover of 60 percent or more are considered closed forests.

Codominance—Refers to a combination of two or more species that share dominance in the same vegetation layer (see Dominance). These species must contribute to a crown cover within plus or minus 10 percent of each other.

Crown cover—The vertical projection of a tree or shrub crown perimeter to the ground.

Cryptogam—A group of primitive plants such as mosses, club mosses, lichens, and ferns that do not produce true flowers or seeds.

D.b.h.—Diameter-at-breast-height. The diameter of a tree trunk at 1.3 m above the ground.

Dicotyledon—Trees of the class Angiospermae whose seeds contain two cotyledons.

Dominance—Refers to the plant species providing the greatest crown cover in any vegetative layer.

Forb—A broad-leaved herbaceous plant.

Graminoid—Narrow leafed monocotyledonous plants of the families Graminaeae, Cyperaceae, and Juncaceae (excluding bamboos); grass-like plants.

Gymnosperm—Trees with seeds usually borne on the upper side of open scales that are often produced in cones. Includes all trees in the class Gymnospermae.

Herbaceous—Herblike or composed of herbs—plants with soft green leaves and no woody tissue.

Monocotyledon—Trees of the class Angiospermae whose seeds contain a single cotyledon.

Overstory—The taller plants within a vegetation type, forming the upper layer of canopy cover.

Physiognomy—The characteristic structure of vegetation, apart from land form.

Shrub—A low-branching woody perennial, usually less than 5 m tall, and often having several main stems arising from a central point in the root system.

Synusia—A structural vegetative unit often made up of several species, characterized by relative uniformity of life form. It may be a layer within the physiognomic profile of a community, or it may be a life form type within a layer (for example, epiphytes growing within an overstory layer).

Tree—A woody plant that usually has an erect perennial stem or trunk at least 7.5 cm d.b.h. and a total height of at least 5 m.

Woodland—A unit of vegetation dominated by trees whose crowns generally are not touching. Plant communities with trees having a crown cover of 25 to 60 percent are considered woodlands.

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A classification system for use in describing vegetation has been developed for Hawaii. Physiognomic and taxonomic criteria are used for a hierarchical stratification of vegetation in which the system categories are Formation, Subformation, Series, Association, and Phase. The System applies to local resource management activities and serves as a framework for resource assessment reporting as it relates to vegetation. Although developed for Hawaii, the system can be applied to other Pacific islands.

Retrieval Terms: vegetation types, plant communities, classification, Hawaii, vegetation classification, tropical forestry, island forestry

United States
Department of
Agriculture

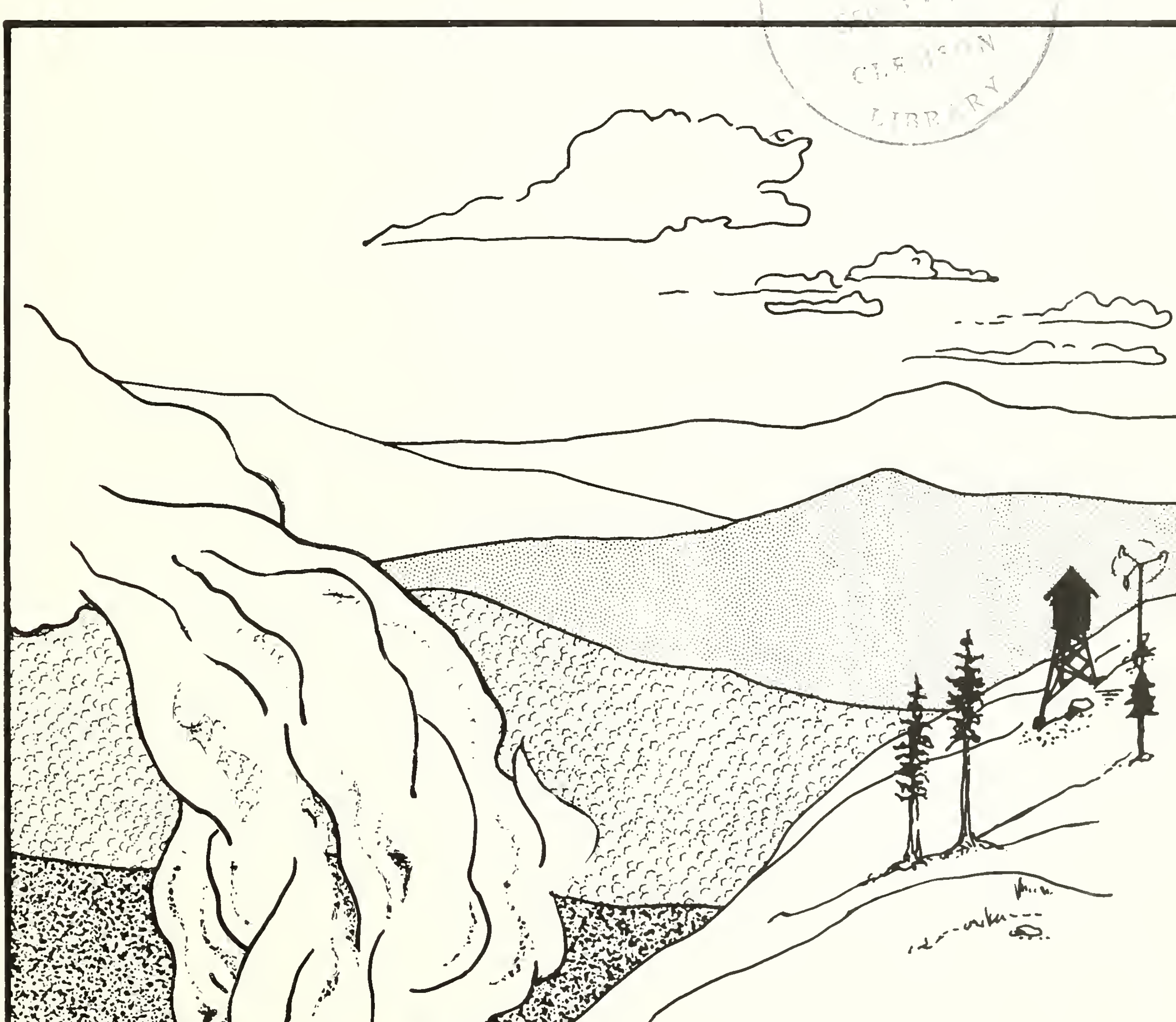
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-77

Potential Fire Behavior in California: an atlas and guide for forest and brushland managers

Bill C. Ryan



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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

July 1984

Potential Fire Behavior in California: an atlas and guide for forest and brushland managers

Bill C. Ryan

CONTENTS

Knowledge of potential fire-behavior characteristics is needed by forest and brushland managers. They need to know the most probable dates or periods of dangerous fire conditions and probable fire intensities and spread rates. For prescribed fires, they need to know probable dates and lengths of time the burns can be continued within prescription limits. They need to know when weather and fuel conditions will allow accumulated debris and fuel to burn, but not burn so intensely that desirable vegetation is killed or the fire becomes wild. They also need information to estimate whether or not the wind will take smoke into populated areas and cause problems.

At the National Fire Weather Data Library, Fort Collins, Colorado, the Forest Service, U.S. Department of Agriculture, maintains on magnetic tapes a mass of data on weather and fuel moisture. These data are collected from fire-danger rating stations throughout the United States (Furman and Brink 1975). With these data and techniques developed by Deeming and others (1977) potential fire characteristics can be estimated.

This report describes an Atlas that provides statistical analyses of potential fire behavior in California's wildlands. Charts and information from the fire-danger rating station at Mount Hebron in northern California serve as an example of the information provided by the Atlas.

THE ATLAS

The Atlas includes 10 volumes. Volume 1 lists the contents and describes the data used to derive the statistics and information in the Atlas. The other nine volumes include one volume for each of the nine sections of California *(fig. 1).*

Maps of each section *(appendix A: figs. 2 to 10; appendix B: tables 1 to 18)* show locations of observation stations whose fire weather statistics are included in the Atlas. The maps are accompanied by a list of the stations in each section showing each station name, number, number of years of observation, and the total number of observations. Stations on the maps are identified by the last four digits of the station number. Because some areas overlap two sections, some station locations are plotted in two sections. When that occurs, information and statistics for that station are included in only one section. The section is indicated by a number next to the station number.

Application

The information contained in the Atlas can help managers in . . .

Prescribed burning:

- Estimating the probable number of days available with spread component (SC), burning index (BI), and ignition component (IC) within prescription limits.
- Determining probable length of time that weather and fuel variables will remain within prescription limits and burns can be continued.
- Estimating potential fire characteristics between observations at danger-rating stations.
- Determining dates or periods when prescription conditions will probably occur.
- Providing data to facilitate estimating number of acres that can be burned within prescription each month.

Fire suppression:

- Estimating probable number of days in, above, or below specific ranges of fire intensity or spread rates.
- Estimating probable maximum fire intensities and spread rates.
- Estimating probability of fire intensities or spread rates within specific ranges.
- Estimating most probable dates or periods of dangerous fire conditions.

General purposes:

- Estimating probability of occurrence of specific weather conditions and fuel moisture.
- Estimating probability of specific ranges of windspeed and direction during a given month.

Because the information in the Atlas is based on the National Fire-Danger Rating System—1978 (NFDRS), the four basic principles of the System should be considered when using the Atlas:

"1. The NFDRS relates only to the potential of the initiating fire. An initiating fire is one that does not behave erratically; it spreads without spotting through continuous

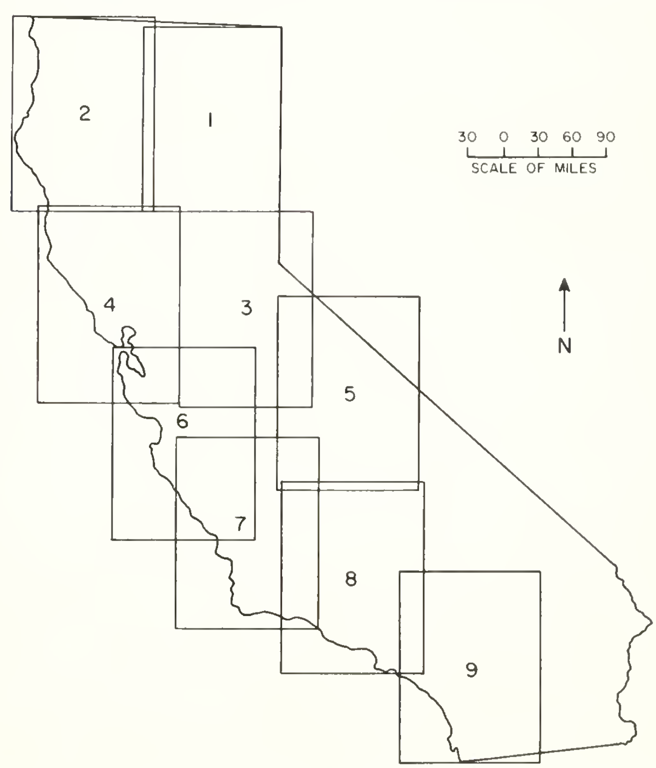

Figure 1—Areas of California are outlined into nine sections. Each section contains fire-danger observation stations, as indicated on maps *(appendix A: figs. 2-10).*

ground fuels. Crowning and spotting are not now addressed. However, experience with the NFDRS will enable users to identify the critical levels of fire danger when such behavior is highly probable.

"2. The System only addresses those aspects of fire control strategy affected by fire occurrence and behavior. The concept of containment, as opposed to extinguishment, is basic because it allows us to limit the scope of the rating problem to the behavior potential of the headfire. Other aspects of the containment job such as accessibility, soil condition, and resistance to line construction must be evaluated by other means.

"3. The ratings are relative, not absolute. Wherever possible, we have structured the component or index so that it is linearly related to the particular aspect of the fire problem being rated. Thus, when the value of a component or index doubles, the fire manager should expect a doubling of the rated activity relative to what has been recently observed. *The BI is an exception that will be addressed later.*

"4. Fire danger is rated from a *worst case* approach. Fire weather is measured at the time of day when fire danger is normally the highest; and, wherever possible, in the open at midslope on southerly or westerly exposures. This important principle must be understood if fire-danger ratings are to be properly interpreted."

Derivation

To develop the Atlas, data from the National Fire Weather Data Library were used. Data were processed from magnetic tapes with FORTRAN code modified from the National Fire-Danger Rating System's (Deeming and others 1977) FIREDAT routine (Main and others 1982). Necessary variables such as 1-hour timelag fuel moisture (TLFM), 10-hour TLFM (if necessary), 100-hour TLFM, 1000-hour TLFM, herbaceous and woody fuel moisture, spread component, burning index, and ignition component were computed.

Spread component, BI, and IC were calculated with current (summer 1981) fuel models, slope, and herbaceous types, as recorded on the 10-day Fire Danger and Fire Weather Record for the stations. The date of greenup—when the spring flush of growth becomes generally apparent—is needed for calculations of live fuel moisture. The dates stored in the Administrative Forest Fire Information Retrieval and Management System (AFFIRMS)—for 1981 and 1982 were used. When the greenup date was not cataloged in AFFIRMS, dates were obtained from the agency responsible for the observations or were estimated on the basis of elevation and dates given for surrounding stations. The potential fire characteristics in or near the month in which greenup begins may vary greatly. If greenup during a particular year is earlier than the date used to produce the tables, fuel moisture will tend to be greater and fire severity potential will tend to be less between the two dates. If greenup is later, fuel moisture will tend to be less and fire severity potential greater.

Information about each observation station's fuel models, location, elevation, length of record, agency affiliation and protection unit are given. The information for Mount Hebron Ranger Station *(fig. 11)* is included as an example, along with statistics of weather and potential fire characteristics *(tables 1-18)*. Inventories of observations each year of record for fire-danger observation stations can also be obtained from the National Fire Weather Data Library (Furman and Brink 1975).

INTERPRETING THE DATA

Fire weather observations, except for minimum and maximum temperatures and relative humidities, are recorded for 1300 P.s.t. only. Statistics derived from observations taken at one time of the day cannot be assumed to be representative of conditions at other times of the day. This fact must be considered when using the statistics.

Length of record of observations taken at a location is significant because the longer the record, the more confidence that can be placed in the statistics of that record. In California, records vary from only a few months to more than 20 years. The confidence that can be placed in the statistics, therefore, varies greatly from station to station. In the Atlas, statistics were computed for only those stations with at least 4 years of data for 1 or more months of the year after 1972. The number of observations and years of record have been included with the other statistics to help users evaluate the reliability of the information and establish confidence levels. For example, in April 1973 through December 1981, at Mount Hebron Ranger Station, the SC, BI, and IC indicated potentially severe fire conditions compared with other months *(tables 2, 4, 6)*. Only 11 observations were recorded in April during the 9-year period, however. Therefore, these statistics are not reliable indicators of potential fire conditions to be expected in April.

Persistence of Potential Fire Characteristics

The persistence of the spread component was analyzed by dividing its range into 10 intervals *(table 1)*. Class intervals were chosen subjectively on the basis of ranges of spread rates for different fuels. The number of consecutive days from the initial day (0) to runs of nine (9) days, was calculated. The percentages of time, according to records, that the SC calculated on day 0 lasted 1, 2, 3, and up to 9 days, were tabulated. The number 29 (circled in *table 1*)

indicates that 29 percent of the time during periods of record when an SC of between 6 and 10 occurred, it remained 6 or more for 4 days. The number of occurrences of SC in each class is in the column labeled "Obs." (for observations). For example, the number of occurrences of SC between 6 and 10 during the period of record was 673. (The SC is scaled so that it is numerically equal to the theoretical rate of spread in feet per minute.)

The persistence of the BI *(table 3)* and IC *(table 5)* were also determined; these tables are similar to *table 1*. Class intervals for BI were based on the adjective classes used by the Pacific Southwest Region and the California Department of Forestry. Those for IC were obtained by dividing the range (0-100) into 10 equal parts.

Persistence of SC, BI, and IC was considered broken if a 3-day or greater break in sequence of observations occurred; that is, if 3 or more days in a row of observations were missing, persistence of all classes (except the lower class) was considered broken. Calculations were made only after 1972 because records of minimum and maximum temperatures and relative humidities were not available for earlier years.

Frequency Distribution

Spread component, BI, IC, fuel moisture variables, and weather variables were divided into 10 classes each. An empirical distribution (survival) function

$$F_m(x) = \frac{N(\{x_i:\ x_i \geq X\})}{n} \cdot 100$$

was calculated for each of the 10 lower class boundaries X for each month in which

$$N(\{x_i:\ x_i \geq X\})$$

is the number of observed values x_i that are greater than or equal to X, and n is the total number of observations for the month. The function was computed to give the frequency of occurrence for each variable for each month in *tables 2, 4, 6, 7, 9,* and *11* through *16*. The total number of observations for each class, for each month and for the total periods of record are given for each variable.

Class intervals for SC were chosen subjectively on the basis of ranges of spread rates for different fuels. Class intervals for BI were determined on the basis of the adjective classes used by the Pacific Southwest Region and the California Department of Forestry. The class intervals for the other variables were obtained by dividing the estimated maximum ranges of the variables into 10 equal parts.

Table 7 gives the empirical distribution function as defined above for temperature for each month. The number circled in *table 7*, for example, indicates that 74 percent of the time in June the temperature at 1300 P.s.t. reaches at least 65° F (18.3° C). The percentage of temperature in each specific class can be found by subtracting the percentage on the line below. For example, the percentage of days with temperatures at 1300 P.s.t. from 65° F (18.3° C) to 77° F (25° C) in June is 43 (74 percent minus 31 percent). The number of days with temperature from 65° F (18.3° C) to 77° F (25° C) in June is 263 (43 percent of 612)—an average of about 12 ½ days each June for the 21-year period.

Maximum and minimum magnitudes and number of observations for each variable for the months of record are also included in the tables.

Statistics of 100-hour TLFM, 1000-hour TLFM, herbaceous-fuel moisture, and woody-fuel moisture content were calculated only for the years after 1972, when records of maximum and minimum temperatures and relative humidities were generally available.

Minimum and Maximum Temperatures and Relative Humidities

Records of maximum and minimum temperatures and relative humidities were not taken until 1973 at fire weather stations. Thus, *tables 8* and *10* are based on data taken after 1972. The record highs, record lows, mean maximums, mean minimums, and means for each month are given in degrees Fahrenheit.

Precipitation

Precipitation amounts and means in inches are given for all months and years of record when possible *(table 17)*. Often, records for months or years are incomplete, so monthly or annual precipitation cannot be determined. For example, no complete year's record was made for Mt. Hebron Ranger Station.

Windspeed and Direction

The percent of joint occurrences of windspeeds and directions for each month were computed (for example, *table 18*). Windspeeds are divided into 10 classes as shown on the left side of the table. Directions are given to eight points of the compass (eight direction classes). The tables list the percent of joint occurrences of windspeeds in 3 mi/h (1.3 m/s) classes, and directions in eight points of the compass. For example, in *table 18*, the circled number 2 indicates that in the 21-year period in the month of May, SE winds from 10 to 12 mi/h (4.5 to 5.4 m/s) occurred 2 percent of the time at 1300 P.s.t. at Mt. Hebron Ranger Station.

Because of their similarity to other tables, several tables included in the Atlas for Mount Hebron are not included here. They are the tables that show joint occurrences of windspeed and direction for Mount Hebron for other months of record, June-November, and tables of SC, BI, and IC for fuel model T, which are similar to *tables 1-6* in this report for fuel model G.

APPENDIX A—Figures 2 to 10

Figures 2 through *10* each contain a map of a section of California. Each map is accompanied by a list of the stations in the section showing station name and number, number of years of observations, and total number of observations. Stations on the maps are identified by the last four digits of the station number. Because some areas overlap two sections, some station locations are plotted in two sections. A number (superscript) after a station number indicates the section in which information and statistics are given for that station. The following abbreviations are used in the lists of stations:

BRKRDG = Breckenridge
FFS = Forest Fire Station
FS = Fire Station
GS = Guard Station
HQ = Headquarters
LAVBDS = Lava Beds
LO = Lookout
MLKRCH = Milk Ranch
RD = Ranger District
RS = Ranger Station
SAWMPK = Saw Mill Peak
WBO = Weather Bureau Office
WHSHQD = Whiskeytown

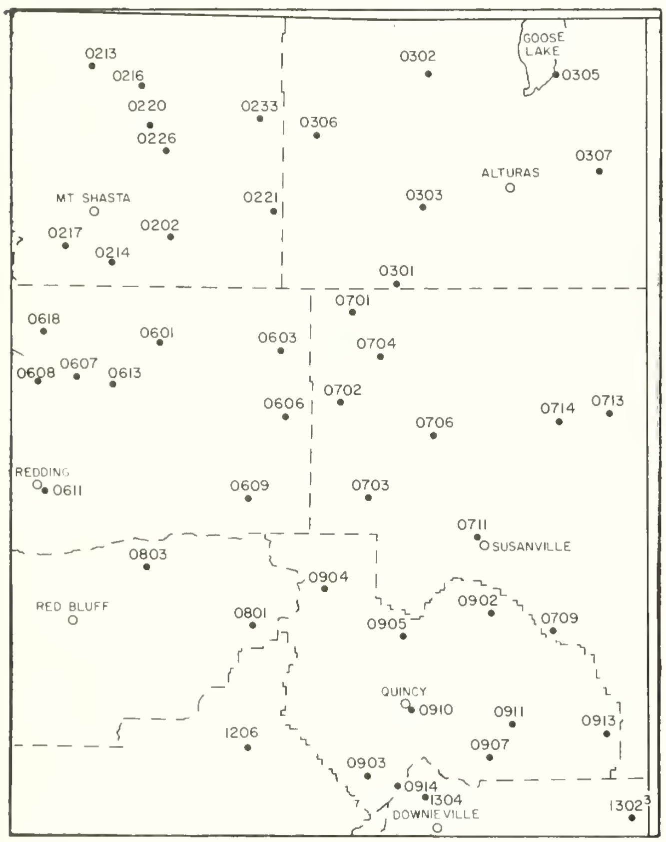

Figure 2—Section 1.

Stations in Section 1:

Station	No.	Years	Observations
Black Fox Mountain LO	40202	19	2460
Lodge Pole GS	40213	12	1213
McCloud RS	40214	9	1453
Mt. Hebron RS	40216	21	3679
Mt. Shasta WBO	40217	20	5248
Orr Mountain LO	40220	18	2390
Round Mountain LO	40221	19	2536
Tennant GS	40226	18	1857
LAVBDS	40233	6	525
Adin RS	40301	11	1755
Blue Mountain LO	40302	18	2120
Canby RS	40303	21	5386
Sugar Hill LO	40305	11	1430
Timber Mountain LO	40306	18	2558
Cedarville	40307	12	1263
Big Bend GS	40601	19	2745
Fall River Mills RS	40603	20	4636
Hat Creek Rim LO	40606	11	1541
Hirz Mountain LO	40607	19	2279
Lakeshore GS	40608	19	2957
Manzanita Lake	40609	19	2487
Redding	40611	6	922
Squaw Creek GS	40613	8	924
Sims	40618	10	1284
Bieber FFS	40701	8	1170
Blacks Ridge LO	40702	19	2512
Bogard RS	40703	20	2939
Boyd Hill LO	40704	9	1150
Dow Butte LO	40706	20	2876
Laufman RS	40709	19	2989
Susanville RS	40711	15	2428
Observation Mountain	40713	12	1352
Ravendale	40714	12	1434
Colby Mountain LO	40801	20	2611
Inskip	40803	6	849
Boulder Creek GS	40902	18	2161
Camel Peak LO	40903	14	1733
Almanor RS	40904	20	3868
Greenville RS	40905	16	2581
Mohawk GS	40907	16	2541
Quincy HQ	40910	19	5091
Smith Peak LO	40911	18	2276
Chilcoot	40913	19	2520
Lexington	40914	5	525
SAWMPK	41206	6	817
Saddleback LO	41304	17	1954

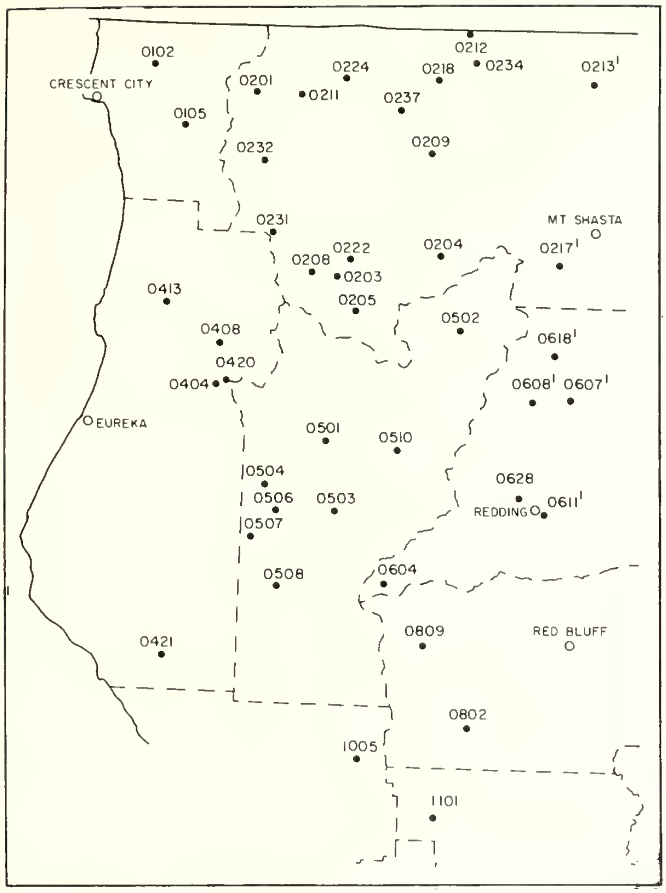

Figure 3—Section 2.

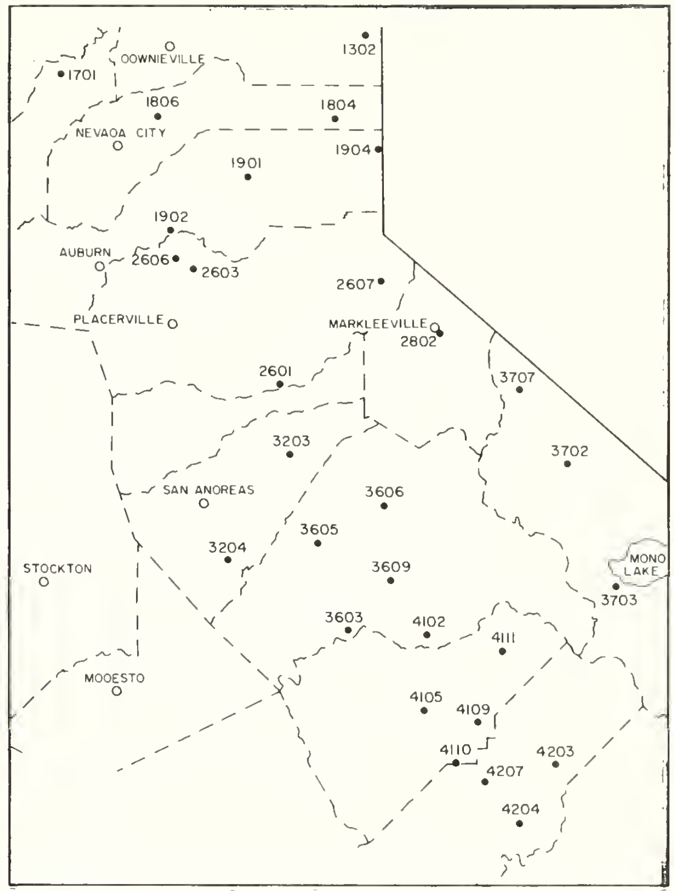

Figure 4—Section 3.

Stations in Section 2:

Station	No.	Years	Observations
Gasquet RS	40102	20	2811
Ship Mountain	40105	7	787
Bald Mountain	40201	5	642
Blue Ridge LO	40203	19	2452
Callahan RS	40204	20	3747
Crawford Creek GS	40205	17	2221
Forks of Salmon GS	40208	17	2288
Fort Jones RS	40209	16	4592
Happy Camp RS	40211	11	1763
Oak Knoll RS	40218	20	4718
Sawyers Bar	40222	21	4288
Seiad RS	40224	18	3761
Somesbar	40231	16	2838
Okonom Lookout	40232	15	1742
Parcgy	40234	6	809
Collins Creek Baldy LO	40237	9	1202
Brush Mountain LO	40404	20	2691
Hoopa	40408	8	1197
Schoolhouse Peak LO	40413	15	1765
Willow Creek	40420	13	2629
Eel River	40421	8	1108
Big Bar RS	40501	20	3310
Coffee Creek RS	40502	20	3110
Hayfork RS	40503	20	3404
Hyampom GS	40504	18	2428
Limedyke LO	40506	19	2714
Mad River RS	40507	20	3062
Ruth RS	40508	18	2301
Weaverville RS	40510	20	5030
Harrison Gulch RS	40604	20	3420
WHSHQD	40628	9	1439
Eagle Peak LO	40802	20	2634
Saddle Camp GS	40809	10	1151
Eel River RS	41005	20	2833
Alder Springs	41101	20	2827

Stations in Section 3:

Station	No.	Years	Observations
Dog Valley	41302	18	2460
Challenge RS	41701	19	2997
Truckee RS	41804	20	4343
White Cloud	41806	7	1025
Duncan Peak GS	41901	18	2059
Forest Hill FS	41902	20	4080
Stateline LO	41904	20	2583
Armstrong Hill LO	42601	18	2491
Bald Mountain LO	42603	19	2892
Georgetown RS	42606	20	5837
Meyers RS	42607	20	4582
Markleeville	42802	18	2831
Blue Mountain LO	43203	14	1792
Fowler	43204	13	1631
Groveland RS	43603	20	5343
Mt. Elizabeth LO	43605	20	2862
Pine Crest RS	43606	20	3091
Woods Ridge LO	43609	17	2281
Bridgeport RS	43702	18	3436
Lee Vining	43703	20	5426
Walker	43707	7	936
Crane	44102	9	1335
Jerseydale FFS	44105	18	3454
Wawona	44109	9	1329
Miami	44110	10	1453
Valley	44111	4	621
Minarets RS	44203	20	2795
North Fork RS	44204	20	6034
Batterson RS	44207	10	1991

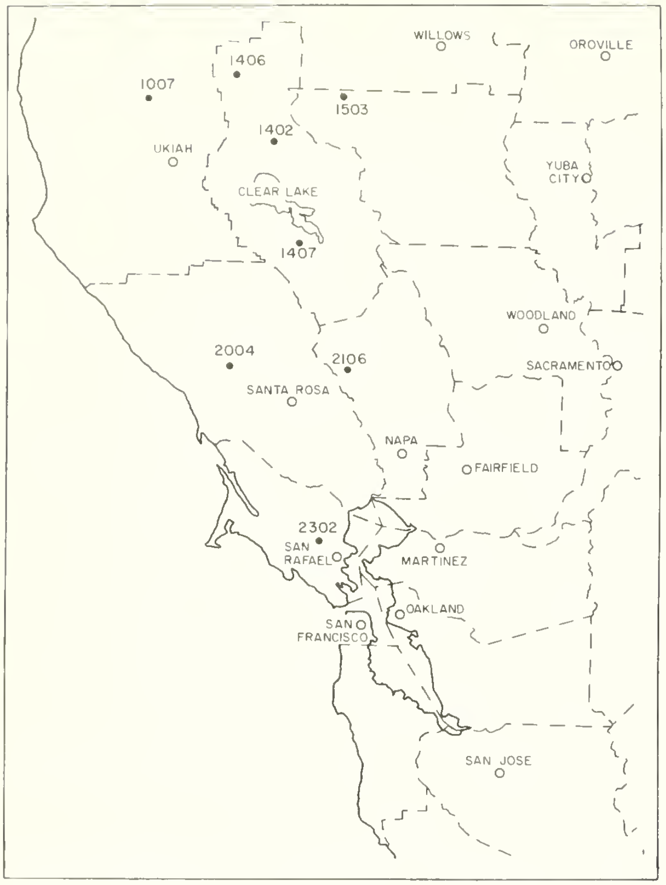

Figure 5—Section 4.

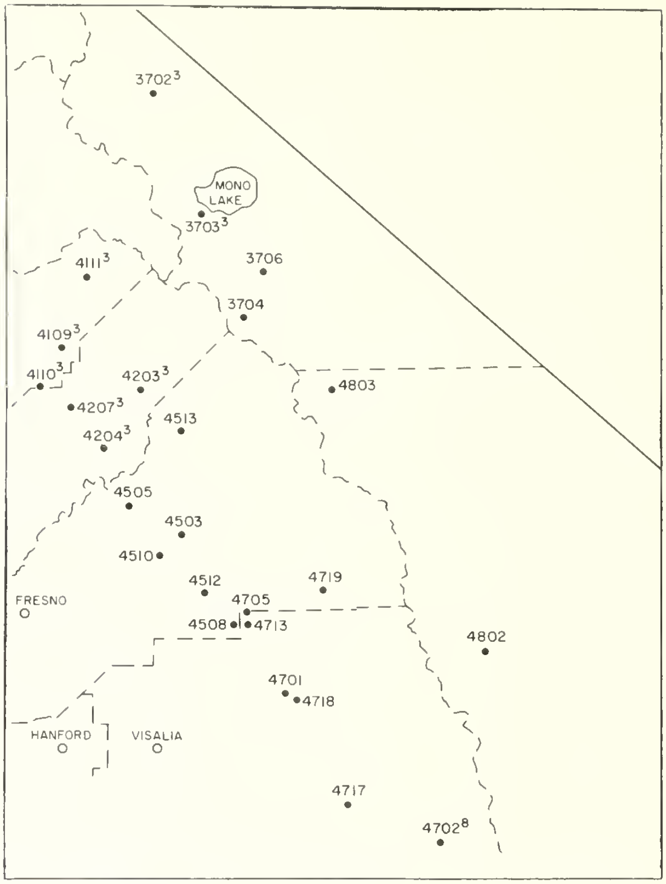

Figure 6—Section 5.

Stations in Section 4:

Station	No.	Years	Observations
Howard Forest FFS	41007	15	1928
High Glade LO	41402	17	2245
Soda Creek GS	41406	20	2918
Konocti	41407	9	1240
Stonyfork RS	41503	20	6066
Mt. Jackson LO	42004	8	992
St. Helena	42106	8	1087
Woodacre FS	42302	8	1056

Stations in Section 5:

Station	No.	Years	Observations
Mammoth RS	43704	20	3831
Bald Mountain LO	43706	8	1039
Fence Meadow GS	44503	20	3068
Mountain Rest	44505	10	1630
Pinehurst RS	44508	20	4076
Trimmer	44510	10	2273
Delilah LO	44512	10	1425
Kaiser	44513	5	573
Ash Mountain	44701	21	3398
Grant	44705	5	669
Park Ridge	44713	14	1605
Wishon	44717	10	1309
MLKRCH	44718	6	731
Cedar	44719	6	892
Lone Pine RS	44802	20	3863
Round Valley	44803	18	4835

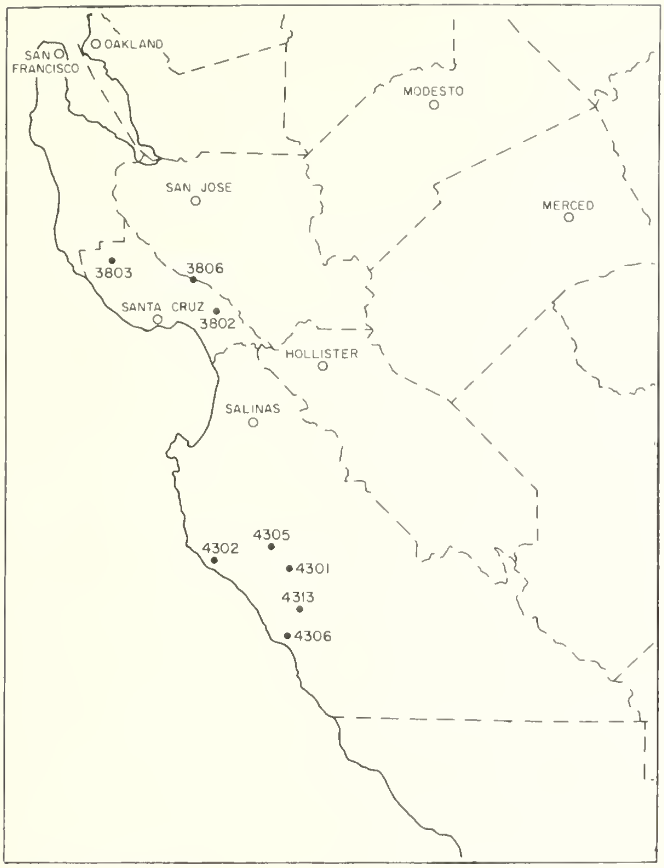

Figure 7—Section 6.

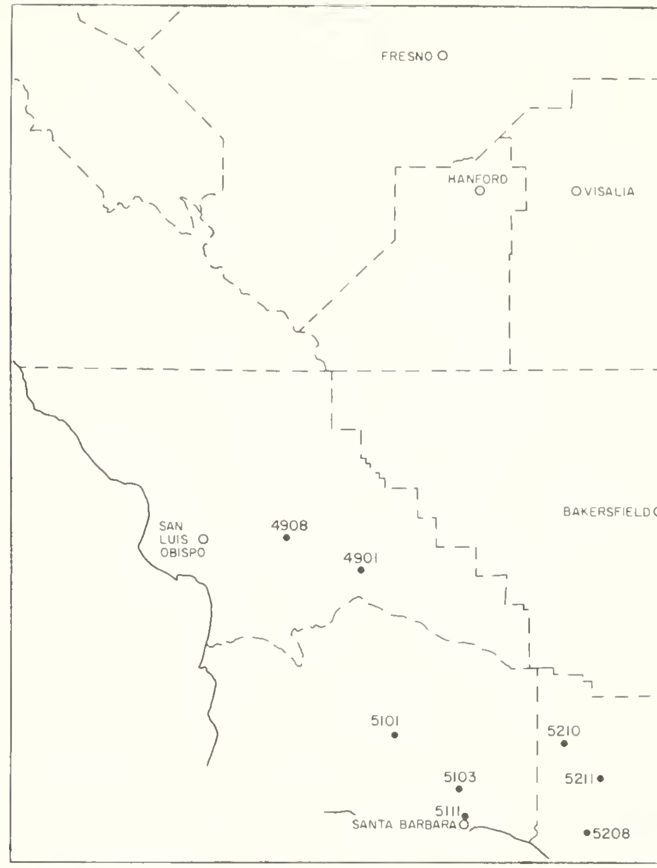

Figure 8—Section 7.

Stations in Section 6:

Station	No.	Years	Observations
Corralitos	43802	6	899
Eagle Rock	43803	13	1807
Burrell	43806	5	550
Arroyo Seco GS	44301	20	5265
Big Sur GS	44302	19	3577
Chews Ridge LO	44305	19	2789
Cone Peak LO	44306	19	2754
The Indians	44313	19	2517

Stations in Section 7:

Station	No.	Years	Observations
Branch Mountain LO	44901	19	2829
Pozo GS	44908	19	3703
Figueroa GS	45101	19	3746
Los Prietos RS	45103	19	5730
Monastery	45111	5	939
Casitas	45208	16	2417
Ozena	45210	10	1718
Rose Valley	45211	9	1613

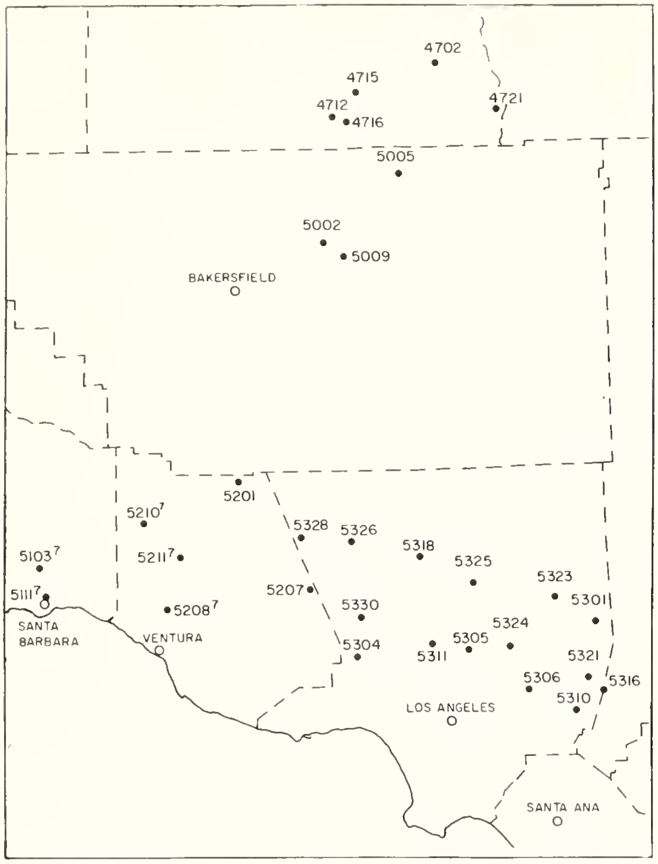

Figure 9—Section 8.

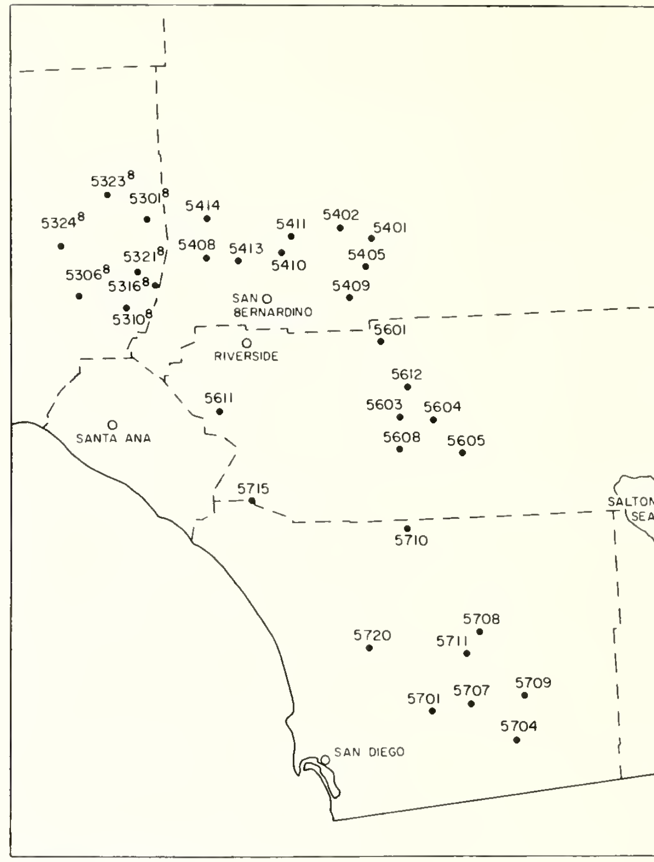

Figure 10—Section 9.

Stations in Section 8:

Station	No.	Years	Observations
Bald Mountain	44702	18	2629
Uhl	44712	16	4346
Camp Whitsett	44715	8	921
Tobias	44716	10	1322
Chimny	44721	4	512
Democrat Springs	45002	19	3050
Kernville FS	45005	20	4286
BRKRDG	45009	5	611
Chuchupate	45201	19	6055
Temescal RS	45207	20	3859
Big Pines	45301	19	4501
Chatsworth FS	45304	8	1338
Clear Creek FS	45305	10	2092
Duarte FS	45306	13	2325
Lechuza FS	45310	8	1166
Little Tujunga GS	45311	19	3913
Padua Hills FS	45316	17	4124
Sierra Pelona LO	45318	19	2982
Tanbark	45321	10	1415
Vallyermo RS	45323	20	4793
Vetter LO	45324	19	2546
Vincent FS	45325	17	4389
Warm Springs LO	45326	20	3401
Slide Mountain	45328	8	994
Newhall	45330	8	1274

Stations in Section 9:

Station	No.	Years	Observations
Fawnskin	45401	20	6162
Big Pine Flats GS	45402	18	2937
Converse GS	45405	19	3646
Lytle Creek RS	45408	20	6304
Mill Creek RS	45409	20	6388
Strawberry Peak LO	45410	19	3362
Rock Camp RS	45411	18	3037
Devore	45413	9	2445
Mormon Rocks	45414	10	1948
Banning GS	45601	18	3394
Cranston GS	45603	19	3645
Keenwild GS	45604	20	5392
Kenworthy GS	45605	19	3241
Red Mountain LO	45608	19	2855
Temescal GS	45611	20	4652
Vista Grande GS	45612	16	2776
Alpine GS	45701	20	4085
Cameron GS	45704	20	4071
Descanso RS	45707	20	6273
Julian FFS	45708	16	3292
Laguna GS	45709	20	3645
Oak Grove RS	45710	20	5732
Pine Hills GS	45711	20	3588
Tenaja GS	45715	20	3819
Ramona	45720	9	2357

APPENDIX B—Figure 11, Tables 1 to 18

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MT HEBRON RS             40216

        AGENCY / PROTECTION UNIT:                       USFS/ KLAMATH

        LOCATION
             COUNTY:                                    SISKIYOU
             LATITUDE / LONGITUDE:                      41.80N/122.00W
             SECTION:                                   32
             TOWNSHIP / RANGE:                          46N / 1W

        ELEVATION:                                      4250 FT.
        CLIMATE CLASS:                                  2
        GREEN-UP DATE USED FOR COMPUTATIONS:*           05/17

        FUEL MODEL 1/SLOPE CLASS / HERB TYPE:           G /2/ P

        FUEL MODEL 2/SLOPE CLASS / HERB TYPE:           T /1/ P

        NUMBER OF YEARS:                                21

        TOTAL NUMBER OF OBSERVATIONS:                   3679

    * THIS GREEN-UP DATE IS EITHER THE DATE STORED IN AFFIRMS
    IN 1981 OR 1982, OBTAINED FROM THE PROTECTION AGENCY,
    OR INTERPOLATED FROM SURROUNDING STATIONS IF NO DATE
    WAS STORED.
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Figure 11—Information for Mount Hebron Ranger Station.

Table 1—*Percent probability of n consecutive days of spread component remaining in or becoming greater than initial range for 1300 P.s.t.*

Mount Hebron RS 40216 Model: G Slope class: 2 Vegetation type: P
Observations: 1374 Years of record: 9

Spread component	Length of run (days)										Obs.
	0	1	2	3	4	5	6	7	8	9	
0-5	100	100	100	100	100	100	100	100	100	100	398
6-10	100	68	51	38	29	22	19	18	16	14	673
11-20	100	34	13	2	1	1	0	0	0	0	242
21-40	100	12	4	0	0	0	0	0	0	0	57
41-80	100	0	0	0	0	0	0	0	0	0	4
81-160	0	0	0	0	0	0	0	0	0	0	0
161-320	0	0	0	0	0	0	0	0	0	0	0
321-640	0	0	0	0	0	0	0	0	0	0	0
641-1000	0	0	0	0	0	0	0	0	0	0	0
1001+	0	0	0	0	0	0	0	0	0	0	0

Table 2—*Cumulative percent probability of spread component occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

Spread component	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-5	—	—	—	100	100	100	100	100	100	100	100	—	398
6-10	—	—	—	100	83	63	72	68	68	80	55	—	673
11-20	—	—	—	73	40	18	18	20	24	17	9	—	242
21-40	—	—	—	55	9	2	1	4	6	5	0	—	57
41-80	—	—	—	9	0	0	0	0	0	1	0	—	4
81-160	—	—	—	0	0	0	0	0	0	0	0	—	0
161-320	—	—	—	0	0	0	0	0	0	0	0	—	0
321-640	—	—	—	0	0	0	0	0	0	0	0	—	0
641-1000	—	—	—	0	0	0	0	0	0	0	0	—	0
1001+	—	—	—	0	0	0	0	0	0	0	0	—	0
Maximum	—	—	—	60	34	31	29	41	39	49	12	—	
Minimum	—	—	—	6	3	0	1	1	0	3	3	—	
Observations	0	0	0	11	149	263	272	271	254	143	11	0	1374

Table 3—*Percent probability of n consecutive days of burning index remaining in or becoming greater than initial range for 1300 P.s.t.*

Mount Hebron RS 40216 Model: G Slope class: 2 Vegetation type: P
Observations: 1374 Years of record: 9

Burning index	Length of run (days)										Obs.
	0	1	2	3	4	5	6	7	8	9	
0-15	100	100	100	100	100	100	100	100	100	100	16
16-30	100	95	90	86	81	81	76	71	71	71	220
31-35	100	74	63	51	45	38	32	27	24	23	220
36-40	100	65	49	36	26	21	19	17	16	15	260
41-50	100	48	33	22	16	11	8	7	6	5	372
51-60	100	29	16	9	5	3	2	1	0	0	171
61-70	100	24	5	2	0	0	0	0	0	0	66
71-80	100	26	0	0	0	0	0	0	0	0	23
81-90	100	5	0	0	0	0	0	0	0	0	19
91+	100	17	0	0	0	0	0	0	0	0	7

Table 4—*Cumulative percent probability of burning index occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

Burning index	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-15	—	—	—	100	100	100	100	100	100	100	100	—	16
16-30	—	—	—	100	99	99	100	97	99	100	100	—	220
31-35	—	—	—	100	81	76	91	86	77	87	55	—	220
36-40	—	—	—	100	62	56	83	71	61	61	36	—	260
41-50	—	—	—	91	44	40	60	59	45	28	9	—	372
51-60	—	—	—	64	19	15	21	33	19	10	0	—	171
61-70	—	—	—	64	5	5	8	13	7	6	0	—	66
71-80	—	—	—	55	1	3	2	6	4	3	0	—	23
81-90	—	—	—	27	0	2	1	3	2	1	0	—	19
91+	—	—	—	18	0	0	0	1	1	0	0	—	7
Maximum	—	—	—	114	78	90	90	94	97	85	45	—	—
Minimum	—	—	—	38	15	0	12	1	0	19	20	—	—
Observations	0	0	0	11	149	263	272	271	254	143	11	0	1374

Table 5—*Percent probability of n consecutive days of ignition component remaining in or becoming greater than initial range for 1300 P.s.t.*

Mount Hebron RS 40216 Model: G Slope class: 2 Vegetation type: P
Observations: 1374 Years of record: 9

Ignition component	Length of run (days)										Obs.
	0	1	2	3	4	5	6	7	8	9	
0-10	100	100	100	100	100	100	100	100	100	100	175
11-20	100	81	65	61	54	48	44	44	40	38	295
21-30	100	70	56	42	35	30	25	21	19	16	456
31-40	100	48	34	21	12	7	5	4	3	3	285
41-50	100	31	13	6	4	1	0	0	0	0	101
51-60	100	16	6	0	0	0	0	0	0	0	34
61-70	100	27	0	0	0	0	0	0	0	0	17
71-80	100	0	0	0	0	0	0	0	0	0	8
81-90	100	0	0	0	0	0	0	0	0	0	3
91-100	100	0	0	0	0	0	0	0	0	0	0

Table 6—*Cumulative percent probability of ignition component occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

Ignition component	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-10	—	—	—	100	100	100	100	100	100	100	100	—	175
11-20	—	—	—	100	80	82	94	89	87	89	64	—	295
21-30	—	—	—	100	48	57	75	73	67	69	18	—	456
31-40	—	—	—	82	19	22	42	40	37	25	0	—	285
41-50	—	—	—	36	5	7	15	18	13	6	0	—	101
51-60	—	—	—	27	2	3	4	8	5	3	0	—	34
61-70	—	—	—	9	1	1	1	4	3	1	0	—	17
71-80	—	—	—	0	0	0	1	1	1	1	0	—	8
81-90	—	—	—	0	0	0	0	0	0	0	0	—	3
91-100	—	—	—	0	0	0	0	0	0	0	0	—	0
Maximum	—	—	—	67	64	84	84	89	78	72	28	—	0
Minimum	—	—	—	22	0	0	0	0	0	0	2	—	0
Observations	0	0	0	11	149	263	272	271	254	143	11	0	1374

Table 7—*Cumulative percent probability of temperature occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 21

Temperature (°F)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
< 0	100	100	100	100	100	100	100	100	100	100	100	100	0
0-12	100	100	100	100	100	100	100	100	100	100	100	100	0
13-25	100	100	100	100	100	100	100	100	100	100	100	100	1
26-38	96	100	100	100	100	100	100	100	100	100	100	100	40
39-51	71	93	83	95	99	100	100	100	100	99	92	97	330
52-64	11	28	42	67	82	96	100	100	97	85	50	10	755
65-77	0	0	4	23	44	74	97	91	78	49	6	0	1257
78-90	0	0	0	0	7	31	69	61	35	7	0	0	1221
91-103	0	0	0	0	0	0	6	4	0	0	0	0	74
104+	0	0	0	0	0	0	0	0	0	0	0	0	0
Maximum	54	63	69	76	85	96	95	97	91	86	68	54	
Minimum	20	35	32	34	33	41	51	45	40	32	33	37	
Observations	28	29	52	113	389	612	644	638	614	421	108	30	3678

Table 8—*Temperature means and record highs and lows for the period of record after 1972*

Mount Hebron RS 40216 Years of record: 9

Temperature (°F)	Month											
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Record high	—	—	—	76	84	92	97	102	92	88	69	—
Mean maximum	—	—	—	66	65	75	83	81	77	70	55	—
Mean	—	—	—	47	49	57	64	61	56	48	32	—
Mean minimum	—	—	—	29	32	39	44	41	36	27	10	—
Record low	—	—	—	14	11	19	13	27	17	9	-4	—
Observations	0	0	0	11	149	263	272	271	254	143	11	0

Table 9—*Cumulative percent probability of relative humidity occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 21

Relative humidity (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-9	100	100	100	100	100	100	100	100	100	100	100	100	20
10-19	100	100	100	99	99	100	99	99	100	100	100	100	640
20-29	100	100	96	79	88	90	75	75	77	85	98	100	1292
30-39	93	93	79	48	60	55	28	35	40	56	88	97	815
40-49	82	86	60	30	33	29	8	14	19	32	70	87	379
50-59	71	59	44	19	19	15	4	8	9	18	52	67	243
60-69	68	38	23	14	8	9	1	4	5	9	28	37	117
70-79	46	28	15	8	5	5	0	2	3	5	18	13	88
80-89	18	10	8	5	2	2	0	1	2	2	8	3	67
90-100	0	0	0	1	1	0	0	0	0	1	3	3	17
Maximum	88	88	84	92	100	100	89	90	100	100	100	93	
Minimum	25	26	14	9	5	8	5	6	6	9	13	28	
Observations	28	29	52	113	389	612	644	638	614	421	108	30	3678

Table 10—*Relative humidity means and record highs and lows for the period of record after 1972*

Mount Hebron RS 40216 Years of record: 9

Relative humidity (pct)	Month											
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Record high	—	—	—	100	100	100	100	100	100	100	100	—
Mean maximum	—	—	—	69	93	91	88	91	94	96	94	—
Mean	—	—	—	43	63	61	57	58	61	61	66	—
Mean minimum	—	—	—	17	33	30	25	26	28	27	37	—
Record low	—	—	—	8	8	8	7	6	5	9	11	—
Observations	0	0	0	11	149	266	273	272	254	142	11	0

Table 11—*Cumulative percent probability of 1-h TLFM occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 21

1-h TFLM (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-1	100	100	100	100	100	100	100	100	100	100	100	100	0
2-3	100	100	100	100	100	100	100	100	100	100	100	100	543
4-5	100	100	98	88	94	93	76	75	78	91	99	100	1487
6-7	86	97	85	49	60	51	22	27	32	52	91	97	718
8-9	79	76	46	27	36	27	7	12	14	26	65	73	338
10-11	57	48	27	18	19	16	3	7	8	14	38	43	192
12-13	32	31	21	12	11	9	1	4	5	8	22	10	101
14-15	21	24	12	8	5	6	1	2	3	5	15	7	36
16-17	18	21	8	5	3	5	0	2	2	4	12	7	37
18+	7	14	2	4	2	4	0	1	1	3	7	7	82
Maximum	26	19	18	26	26	28	21	26	28	31	32	26	
Minimum	5	5	3	2	2	2	2	2	2	2	3	5	
Observations	28	29	52	113	389	582	613	605	584	401	108	30	3534

Table 12—*Cumulative percent probability of 10-h TLFM occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 21

10-h TLFM (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-1	100	100	100	100	100	100	100	100	100	100	100	100	0
2-3	100	100	100	100	100	100	100	100	100	100	100	100	47
4-5	100	100	100	100	99	100	98	97	99	99	100	100	766
6-7	100	100	96	98	94	92	65	59	64	86	98	100	937
8-9	79	100	88	65	78	64	27	28	37	64	97	100	599
10-11	61	90	83	40	51	41	15	18	21	42	88	97	435
12-13	39	76	44	27	32	26	7	10	12	27	69	77	271
14-15	39	55	27	14	22	17	3	7	8	17	41	60	147
16-17	39	45	17	10	15	13	2	5	6	10	30	33	94
18+	36	31	10	8	11	10	1	4	5	6	19	10	238
Maximum	50	37	32	50	46	50	35	50	50	50	50	38	
Minimum	6	8	4	5	3	3	3	2	2	3	5	9	
Observations	28	29	52	113	389	582	613	605	584	401	108	30	3534

Table 13—*Cumulative percent probability of 100-h TLFM occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

100-h TLFM (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-1	—	—	—	100	100	100	100	100	100	100	100	100	0
2-3	—	—	—	100	100	100	100	100	100	100	100	100	0
4-5	—	—	—	100	100	100	100	100	100	100	100	100	0
6-7	—	—	—	100	100	100	100	100	100	100	100	100	11
8-9	—	—	—	91	100	100	99	98	100	100	100	100	89
10-11	—	—	—	64	100	93	88	84	100	100	100	—	177
12-13	—	—	—	27	95	76	62	71	95	100	100	—	330
14-15	—	—	—	0	70	41	24	50	83	97	100	—	364
16-17	—	—	—	0	31	17	4	20	46	85	100	—	246
18+	—	—	—	0	14	5	1	7	19	36	64	—	162
Maximum	—	—	—	12	25	22	20	36	32	23	20	—	
Minimum	—	—	—	7	10	7	7	7	9	12	16	—	
Observations	0	0	0	11	149	266	273	272	254	143	11	0	1379

Table 14—*Cumulative percent probability of 1000-h TLFM occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

1000-h TLFM (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-1	—	—	—	100	100	100	100	100	100	100	100	—	0
2-3	—	—	—	100	100	100	100	100	100	100	100	—	0
4-5	—	—	—	100	100	100	100	100	100	100	100	—	0
6-7	—	—	—	100	100	100	100	100	100	100	100	—	0
8-9	—	—	—	100	100	100	100	100	100	100	100	—	0
10-11	—	—	—	100	100	100	100	100	100	100	100	—	112
12-13	—	—	—	100	100	100	93	66	100	100	100	—	285
14-15	—	—	—	100	100	88	50	38	76	100	100	—	359
16-17	—	—	—	100	99	52	12	17	48	79	100	—	272
18+	—	—	—	36	68	26	3	10	28	42	91	—	351
Maximum	—	—	—	19	22	20	18	28	25	22	18	—	
Minimum	—	—	—	16	15	11	11	10	12	15	17		
Observations	0	0	0	11	149	266	273	272	254	143	11	0	1379

Table 15—*Cumulative percent probability of woody-fuel moisture occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

Moisture (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-29	—	—	—	100	100	100	100	100	100	100	100	—	0
30-59	—	—	—	100	100	100	100	100	100	100	100	—	0
60-89	—	—	—	100	100	100	100	100	100	100	100	—	443
90-119	—	—	—	0	46	99	93	66	64	5	0	—	580
120-149	—	—	—	0	17	57	14	19	33	5	0	—	299
150-179	—	—	—	0	3	3	0	7	10	0	0	—	39
180-209	—	—	—	0	0	0	0	6	0	0	0	—	18
210-239	—	—	—	0	0	0	0	0	0	0	0	—	0
240-269	—	—	—	0	0	0	0	0	0	0	0	—	0
270+	—	—	—	0	0	0	0	0	0	0	0	—	0
Maximum	—	—	—	60	160	158	145	200	180	140	60	—	
Minimum	—	—	—	60	60	89	82	74	60	60	60	—	
Observations	0	0	0	11	149	266	273	272	254	143	11	0	1379

Table 16—*Cumulative percent probability of herbaceous-fuel moisture occurrence by class and month and minimum and maximum by month for 1300 P.s.t.*

Mount Hebron RS 40216 Years of record: 9

Moisture (pct)	Month												Obs.
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	
0-29	—	—	—	100	100	100	100	100	100	100	100	—	308
30-59	—	—	—	0	61	100	100	100	64	5	0	—	25
60-89	—	—	—	0	45	100	100	100	64	5	0	—	319
90-119	—	—	—	0	29	100	78	38	39	5	0	—	529
120-149	—	—	—	0	15	53	1	7	6	0	0	—	141
150-179	—	—	—	0	5	11	0	6	1	0	0	—	56
180-209	—	—	—	0	1	0	0	0	0	0	0	—	1
210-239	—	—	—	0	0	0	0	0	0	0	0	—	0
240-269	—	—	—	0	0	0	0	0	0	0	0	—	0
270+	—	—	—	0	0	0	0	0	0	0	0	—	0
Maximum	—	—	—	9	180	176	121	150	150	100	15	—	
Minimum	—	—	—	3	4	89	73	59	3	2	4		
Observations	0	0	0	11	149	266	273	272	254	143	11	0	1379

Table 17—*Precipitation (inches)*

Mount Hebron RS 40216

Year	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	Total
	inches												
1961	—	—	—	—	—	1.65	0.14	—	0.69	—	—	—	—
1962	—	—	—	—	—	0.56	—	0.53	0.30	—	—	—	—
1963	—	—	—	—	—	0.78	0.31	0.25	0.08	—	—	—	—
1964	—	—	—	—	—	—	0.33	0.11	0.12	0.10	—	—	—
1965	—	—	—	—	—	1.63	0.26	—	0.02	0.00	—	—	—
1966	—	—	—	—	1.00	0.69	0.11	0.93	0.55	0.17	—	—	—
1967	—	—	—	—	—	2.02	0.01	0.00	0.16	0.88	—	—	—
1968	—	—	—	—	—	0.83	0.00	—	0.08	—	—	—	—
1969	—	—	—	—	—	1.14	0.33	0.00	0.17	1.65	—	—	—
1970	—	—	—	—	—	1.52	0.22	0.00	0.00	—	—	—	—
1971	—	—	—	—	—	—	0.07	0.11	0.64	—	—	—	—
1972	—	—	—	—	—	0.39	0.02	0.13	0.47	—	—	—	—
1973	—	—	—	—	—	0.00	1.06	0.11	—	—	—	—	—
1974	—	—	—	—	—	0.05	0.13	—	0.00	—	—	—	—
1975	—	—	—	—	—	—	—	—	—	—	—	—	—
1976	—	—	—	—	—	0.77	0.78	3.05	—	—	—	—	—
1977	—	—	—	—	—	1.24	0.07	—	—	—	—	—	—
1978	—	—	—	—	—	—	0.58	—	—	—	—	—	—
1979	—	—	—	—	—	0.25	0.13	1.40	—	—	—	—	—
1980	—	—	—	—	—	1.30	—	0.00	0.48	—	—	—	—
1981	—	—	—	—	—	0.22	—	0.18	—	—	—	—	—
Mean	—	—	—	—	1.00	0.88	0.27	0.49	0.27	0.56	—	—	—

Table 18—*Percent probability of joint occurrence of windspeed and direction at 1300 P.s.t.*

Mount Hebron RS 40216 Total observations: 149
Years of record: 21 May Calm: 0 percent

Windspeed (mi/h)	Wind direction NE	E	SE	S	SW	W	NW	N	Total (pct)[1]	Observations
1-3	0	0	1	0	1	0	0	1	3	5
4-6	3	0	1	3	3	1	7	9	28	41
7-9	3	0	0	1	4	2	9	5	25	37
10-12	0	0	2	1	1	1	6	3	15	22
13-15	1	0	0	1	5	1	5	1	15	22
16-18	0	0	0	1	4	1	4	1	11	16
19-21	1	0	0	0	0	0	1	0	2	3
22-24	0	0	0	0	0	0	1	0	1	1
25-27	0	0	0	1	0	0	1	0	1	2
28+	0	0	0	0	0	0	0	0	0	0
Total percent[1]	9	0	5	7	18	6	34	21	100	
Observations	13	0	7	11	27	9	51	31		

[1]Total percentages may not agree with sums of individual percentages because of rounding.

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Ryan, Bill C. **Potential fire behavior in California: an atlas and guide for forest and brushland managers.** Gen. Tech. Rep. PSW-77. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1984. 15 p.

Potential fire characteristics can be estimated as functions of weather, fuel, and terrain slope. Such information is needed by forest and other land managers—especially for anticipating fire suppression needs and planning prescribed burns. To provide this information, an Atlas has been developed for California. The Atlas includes statistical analyses of spread component, burning index, ignition component, temperature, relative humidity, dead fuel moisture, live woody fuel moisture, live herbaceous fuel moisture, precipitation, windspeed and direction for 200 fire-danger rating stations in California. Charts and information for one of the stations included in the Atlas—Mount Hebron in northern California—serve as an example of this application.

Retrieval Terms: fire management, prescribed burning, fuel moisture, forest climatology, potential fire characteristics

United States
Department of
Agriculture

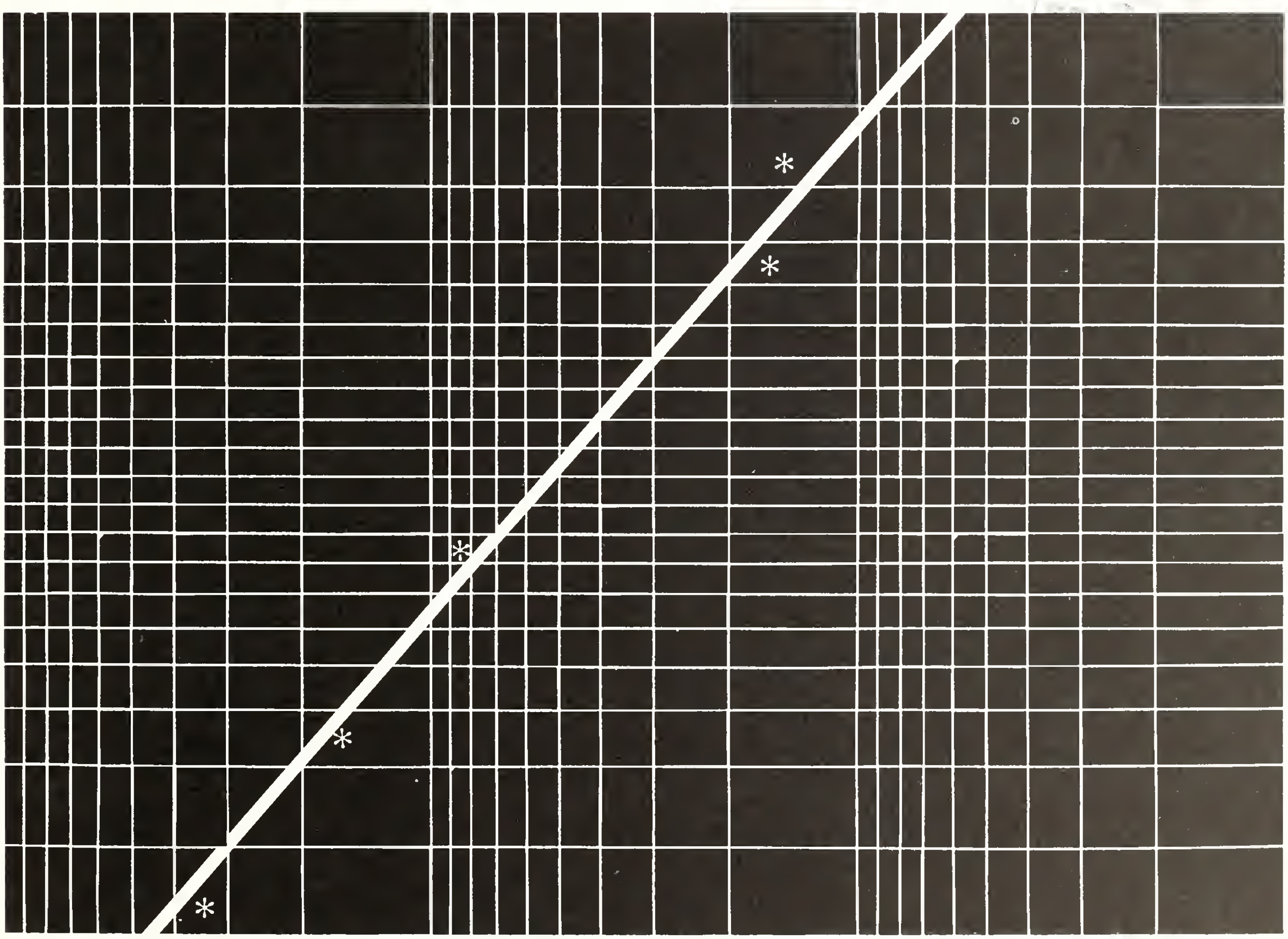
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-78

DOSESCREEN: a computer program to aid dose placement

Kimberly C. Smith
Jacqueline L. Robertson



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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

August 1984

DOSESCREEN: a computer program to aid dose placement

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CONTENTS

The most common method used to evaluate the response of insects to chemicals, such as insecticides, is the dose-response bioassay. An important aspect of designing a chemical bioassay is dose placement—selecting doses of the chemical for testing to obtain effective dose (ED) estimates of high precision.

The placement of doses and allocation of experimental subjects to doses can substantially affect the precision with which an ED is estimated (Smith and others [in press]). The problem of determining the optimal design for a fixed total sample size has been addressed to some extent in the statistical literature (e.g., Abdelbasit and Plackett 1983, Brown 1966, Freeman 1970, Tsutakawa 1980); but the most readily accessible guidelines for bioassay design are probably those by Finney (1971). Finney's guidelines are applicable for estimating ED's in the vicinity of the 50 percent response level.

Research with chemicals often requires precise estimation of an ED at the extremes of the response curve, rather than in the middle. For example, derivation of multiplication factors to predict mortality rates in the field on the basis of laboratory data necessitates estimating the ED_{90} or ED_{95} (Haverty and Robertson 1982). Assessment of the toxicity of an insecticide to an endangered or threatened species, on the other hand, entails estimation of ED's at the low extremes of the response curve, such as the ED_{10}. Although it is intuitively obvious that the optimal designs for estimating the ED_{10} and ED_{90} will differ from each other and from the optimal design for the ED_{50}, guidelines for designing experiments expressly for the purpose of estimating an extreme quantile are generally lacking.

DOSESCREEN is a computer program written to assist investigators in dose placement. It is a computationally simple, yet flexible, tool with which to plan experiments that involve a binary quantal response model with one independent variable. A measure of precision for an estimator of an ED is produced on the basis of the asymptotic expected length of the confidence interval for the ED with a proposed experimental design. The measure represents a generalization of the measure derived by Finney (1971); DOSESCREEN can provide an estimate of precision for any ED calculated with any of a large class of tolerance distributions.

On the basis of the DOSESCREEN measure of precision, an efficient design can be selected for estimating an ED of interest. If two or more ED's are to be estimated from the same experiment, DOSESCREEN can help the investigator find a good compromise design, i.e., one that results in reasonably high precision for estimates of all the ED's of interest. DOSESCREEN output may also provide assistance in selecting the total sample size for an experiment.

Although the language adopted in this report is oriented to entomological bioassays, DOSESCREEN can be used to design experiments in other scientific disciplines in which the levels of the independent variable are controlled by the experimenter. In agricultural research, for example, nursery studies of the effect of fertilizers or of water stress on seedling survival might be improved by careful consideration of treatment level placement. Clinical trials in medicine provide another example, and it is easy to conceive of analogous experimental situations in the behavioral sciences.

This report describes the statistical basis of the DOSESCREEN measure of precision, suggests uses of it, and presents a hypothetical experiment designed on the basis of program output. The DOSESCREEN computer program listed in the *appendix* is written in Fortran 77 so that users can easily convert DOSESCREEN to their own computer system.

1. STATISTICAL BASIS FOR DOSESCREEN

1.1 Binary Quantal Response Models

A dose-response bioassay typically involves selecting T dose levels of a chemical, administering the t^{th} dose level x_t to n_t test subjects, and denoting the response of each subject as either 1 (for example, dead) or 0 (for example, alive). The numbers of subjects per dose level, n_t, are called the *cell sizes* for the experiment. The total number of subjects, $N = \Sigma n_t$, is called the *sample size*.

The statistical framework for this type of experiment is the binary quantal response model with one independent variable. Subjects that receive the same dose are assumed to have the same probability of responding, and probability of response is assumed to be functionally related to the dose level in the form

$$P_t = F\left(\beta_1 + \beta_2 x_t\right)$$

where P_t is the probability of response, x_t is the dose level, and $F(\cdot)$ is a cumulative distribution function (CDF) with density $f(\cdot)$ and inverse $F^{-1}(\cdot)$. The CDF's most commonly used by biologists are the Gaussian

$$F(x) = \int_{-\infty}^{x} e^{-z^2}/\sqrt{2\pi}dz$$

which results in a model traditionally called the probit model, and the logistic

$$F(x) = 1 / (1 + e^{-x})$$

which results in the logit model.

A number of procedures that provide estimators b_1, b_2 of the regression coefficients β_1, β_2 are available. Those most widely used belong to the class of regular best asymptotically normal (RBAN) estimators; these estimators have sampling distributions that approach the same bivariate normal distribution as all the cell sizes approach infinity. For example, the maximum likelihood and minimum logit chi-square estimators are both RBAN estimators. For large n_t,

$$b_1, b_2 \sim N\left(\beta_1, \beta_2, \nu_{11}, \nu_{22}, \nu_{12}\right)$$

where

$$\begin{bmatrix} \nu_{11} & \nu_{12} \\ \nu_{12} & \nu_{22} \end{bmatrix} = \begin{bmatrix} \Sigma n_t w_t & \Sigma n_t w_t x_t \\ \Sigma n_t w_t x_t & \Sigma n_t w_t x_t^2 \end{bmatrix}^{-1} \qquad (1)$$

with $w_t = [f(\beta_1 + \beta_2 x_t)]^2 / P_t(1-P_t)$. Classical inference is based on the use of an RBAN estimator with cell sizes large enough to ensure that the normal distribution provides a good approximation to the distribution of b_1, b_2.

1.2 Effective Doses (ED's) and Confidence Intervals

Let μ_0 denote the dose level that produces a certain probability of response P_0, where $0 < P_0 < 1$:

$$\mu_0 = \left[F^{-1}(P_0) - \beta_1\right] / \beta_2.$$

For example, if $P_0 = 0.5$, then μ_0 is the ED_{50}. An estimate m_0 of μ_0 is obtained by substituting b_1 and b_2 for β_1 and β_2 in the expression for μ_0,

$$m_0 = \left[F^{-1}(P_0) - b_1\right] / b_2.$$

For large samples, the $(1-\alpha)100$ percent confidence limits for μ_0 (Finney 1971) are

$$\frac{m_0 + g\left(m_0 + v_{12}/v_{22}\right)}{1-g} \pm \frac{z\sqrt{v_{11} + 2m_0 v_{12} + m_0^2 v_{22} - g(v_{11} - v_{12}^2/v_{22})}}{b_2(1-g)}$$

where v_{ij}, $i,j = 1,2$ are the estimated variances and covariance of b_1, b_2 computed by substitution of the estimated parameters for their true values in (1), z is the $(1-\alpha/2)100$ percent quantile of the normal (0,1) distribution, and $g = z^2 v_{22} / b_2^2$.

1.3 DOSESCREEN Measure of Precision

The length of the confidence interval for μ_0 is

$$L = \frac{2z\sqrt{v_{11} + 2m_0 v_{12} + m_0^2 v_{22} - g(v_{11} - v_{12}^2/v_{22})}}{b_2(1-g)} \quad \text{if } g<1 \qquad (2)$$

$$= \infty \quad \text{if } g \geq 1.$$

For any finite sample size, the expected ("average") value of L is infinite, because L is infinite with positive probability; however, as all cell sizes approach infinity, the probability that $g \geq 1$ approaches zero if $\beta_2 \neq 0$, so that asymptotically L has finite expectation E(L). A first order approximation to E(L), denoted by L^*/β_2, is found by replacing all the random variables in (2) by their asymptotic expected values; e.g., b_i is replaced by β_i, v_{ij} by ν_{ij}, and m_0 by μ_0.

L^*/β_2 appears to be a complicated function of the unknown parameters and the dose levels. However, simple algebra shows that L^* is actually a function only of the underlying probabilities of response P_t. For this reason, L^* rather than L^*/β_2 was chosen as the DOSESCREEN measure of precision:

$$L^* = \frac{2z\sqrt{\nu_{11}^* + 2F^{-1}(P_0)\nu_{12}^* + [F^{-1}(P_0)]^2 \nu_{22}^* - g^*(\nu_{11}^* - \nu_{12}^{*2}/\nu_{22}^*)}}{1-g^*}$$

where

$$\begin{bmatrix} \nu_{11}^* & \nu_{12}^* \\ \nu_{12}^* & \nu_{22}^* \end{bmatrix} = \begin{bmatrix} \Sigma n_t w_t^* & \Sigma n_t w_t^* F^{-1}(P_t) \\ \Sigma n_t w_t^* F^{-1}(P_t) & \Sigma n_t w_t^* [F^{-1}(P_t)]^2 \end{bmatrix}^{-1}$$

with $w_t^* = f\left(F^{-1}(P_t)\right)^2 / P_t(1-P_t)$, and $g^* = z^2 \nu_{22}^*$.

When applied to the ED_{50}, L* is related to Finney's (1971, p. 142) measure of precision, Ib^2N, by $Ib^2N = (\Sigma n_t)L^{*2}/4$. L* is not an approximation to confidence interval length unless $\beta_2 = 1$, but is a measure of precision suitable for evaluating the relative performance to be expected from candidate designs for the same experiment, because for any one experiment β_2 will be a constant, irrespective of the design.

1.4 Factors Affecting Accuracy of Approximation

The approximation L* predicts the relative precision of an experimental design over repeated identical experiments. The result of any one experiment will unavoidably depart from the prediction because of random variation alone. However, other factors which are largely under the control of the investigator can introduce error into L*. These are sample size, choice of model, and dose selection.

L* is based on large sample approximations to the sampling distributions of b_1 and b_2 and cannot be expected to be reliable for small samples. A Monte Carlo experiment was performed to evaluate the accuracy of L* using a logit model with $\beta_1 = 0$, $\beta_2 = 1$, and selected designs (Smith and others [in press]). The average and median lengths of confidence intervals produced in 3000 replications using the maximum likelihood estimator were compared with those predicted by L* (*table 1*). The results indicated that L* underestimates the average length in small samples, but is a highly accurate estimator of median interval length.

The recommended placement of doses will depend on the underlying model assumed, that is, on the choice of F. Use of an inappropriate model will introduce bias in L*. Our experience has been that optimal designs for the probit and logit models are virtually identical, but different models generally result in different optimal designs, particularly for estimating an extreme ED.

Accuracy of the approximation also depends on an accurate determination of the dose levels x_t corresponding to the P_t selected. In practice some error will always be introduced in moving from P_t to x_t (Abdelbasit and Plackett 1983). The error can be minimized by conducting preliminary experiments to establish a tentative dose-response relationship for the chemical under investigation.

2. SUGGESTED USES OF DOSESCREEN

DOSESCREEN computes a measure of precision L* for any combination of cell sizes (n_t), response probabilities (P_t), ED, and desired significance level (α). The program is particularly well-suited for use interactively; in this mode, an investigator can explore how precision is affected by sample size, number of dose levels and dose placement, and allocation of sample size to dose levels. DOSESCREEN may be used in other ways as well.

Table 1—*DOSESCREEN (L*) predicted average 95 percent confidence interval length and average and median lengths obtained from Monte Carlo (M.C.) simulation with a logit model*[1]

Response probabilities	Cell size	ED_{50}					ED_{90}				
		L*	M.C.[2] avg.	(Error)	M.C. median	(Error)	L*	M.C. avg.	(Error)	M.C. median	(Error)
				pct		*pct*			*pct*		*pct*
0.20,0.35,0.50,0.65,0.80	24	0.885	0.968	(-8.2)	0.888	(-0.3)	2.635	3.272	(-20)	2.591	(1.7)
	48	.590	.605	(-2.6)	.590	(-.1)	1.665	1.763	(-5.6)	1.647	(1.1)
	96	.406	.411	(-1.3)	.406	(-.2)	1.118	1.148	(-2.6)	1.113	(.5)
0.10,0.80,0.85,0.90,0.95	24	1.492	1.547	(-3.6)	1.497	(-.3)	1.461	1.494	(-2.2)	1.459	(.1)
	48	1.008	1.016	(-.8)	1.006	(.2)	.988	.992	(-.4)	.989	(-.2)
	96	.698	.700	(-.4)	.696	(.2)	.684	.687	(-.4)	.685	(-.1)
0.20,0.30,0.40,0.45	15	.894	1.050	(-15)	.899	(-.6)	3.084	4.574	(-33)	2.997	(2.9)
0.55,0.60,0.70,0.80	30	.585	.609	(-4.0)	.587	(-.4)	1.879	2.052	(-8.4)	1.859	(1.1)
	60	.399	.405	(-1.6)	.400	(-.2)	1.244	1.287	(-3.4)	1.236	(.6)
0.10,0.15,0.70,0.75	15	1.196	1.207	(-1.2)	1.182	(.9)	1.557	1.601	(-2.8)	1.552	(.3)
0.80,0.85,0.90,0.95	30	.811	.813	(-.2)	.806	(.6)	1.048	1.056	(-.7)	1.038	(1.0)
	60	.563	.564	(-.2)	.561	(.4)	.724	.728	(-.6)	.722	(.3)

[1]Regression coefficients, $\beta_1 = 0$, $\beta_2 = 1$, were fixed.

[2]Average length of confidence interval excluded samples that resulted in infinite confidence intervals. Results are for the maximum likelihood estimator in 3000 replications.

2.1 Determining Optimal Design

The DOSESCREEN subroutine can be easily inserted into a computer routine that computes L* for all possible values of P_t, n_t: $\Sigma n_t = N$ to find the optimal placement of doses for a given model, fixed total sample size (N), number of doses (T), and ED. Because of computer time constraints, in our computer routine we restricted the search to values of P_t that are multiples of 0.05 and required equal numbers of subjects per dose, so that $n_t = N/T$. Optimal designs subject to these constraints are reported in *table 2* for estimation of the ED_{50} and ED_{90}, assuming a logit model. Because of symmetry, the optimal design for estimation of the ED_{10} can be found from that of the ED_{90} by translation of the P_t. For example, optimal P_t for the ED_{10} at N = 240, T = 3 is (0.15, 0.20, 0.95).

The recommended designs do not change much as N increases. For estimating the ED_{50}, doses should be placed symmetrically on the response curve about the ED_{50}. For increasing N, the doses should be placed somewhat closer together. For precise estimates of the ED_{90}, the majority of the doses should be located in the vicinity of the ED_{90}, with one or two doses located in the region of low response. On the basis of L*, confidence interval length apparently is relatively insensitive to the number of dose levels chosen.

2.2 Finding Compromise Designs

Often an investigator wishes to estimate simultaneously more than one ED from one experiment. For example, in insecticide research, the ED_{90} is often of primary interest, but in the literature the ED_{50} estimate is always reported as well. Since dramatically different designs are recommended for the two ED's, a third design may be desired that will result in moderately high precision for both the ED_{50} and the ED_{90}.

The global search for the optimal design can be modified easily to provide the information necessary to select a good compromise design. Our search program divides all possible designs into categories with similar values of L* for the ED_{50}. Designs in each category are searched separately to find the optimal design for the ED_{90}.

An example of this technique is presented in *table 3.* Because $NL^{*2}/4$ remains relatively stable and of moderate size as N varies, $NL^{*2}/4$ was computed instead of L*. In the example in *table 3* and in all others we examined, the optimal ED_{50} design poorly estimated the ED_{90}, whereas the optimal design for the ED_{90} estimated the ED_{50} rather

Table 3—*Optimal designs for estimating the ED_{90}, by categories of the ED_{50}*[1]

Categories of $NL^{*2}/4$ for the ED_{50}[2]	Optimal design (P_t) within category for the ED_{90}	Value of $NL^{*2}/4$ for optimal design	
		ED_{50}	ED_{90}
Optimal for ED_{50}	0.30,0.35,0.40,0.70,0.75	20.36	213.06
20-30	.15, .30, .80, .85, .90	29.84	74.98
30-40	.15, .65, .80, .85, .90	39.25	67.04
40-50	.15, .75, .85, .90, .95	49.59	61.18
>50	.10, .80, .85, .90, .95	60.98	58.36

[1]Based on a logit model with five doses, a total sample size of N = 240, equal cell sizes, and a level of significance α = 0.05. Designs were considered for which all P_t were multiples of 0.05.

[2]L* = DOSESCREEN measure of precision.

Table 2—*Optimal designs for estimating the ED_{50} and ED_{90} for a logit model* [1]

Sample size	No. doses	Cell size	ED_{50} Optimal P_t	ED_{50} L*	ED_{90} Optimal P_t	ED_{90} L*
120	3	40	0.25,0.50,0.75	0.88	0.05,0.85,0.90	1.51
	5	24	.25, .30, .50,0.70,0.75	.88	.10, .80, .85,0.90,0.95	1.46
	6	20	.25, .30, .35, .65, .70,0.75	.88	.10, .75, .80, .85, .90,0.95	1.49
	8	15	.20, .25, .35, .40 .60, .65, .75, .80	.88	.05, .10, .70, .75 .80, .85, .90, .95	1.54
240	3	80	.30, .45, .75	.58	.05, .80, .85	1.02
	5	48	.30, .35, .45, .70, .75	.58	.10, .80, .85, .90, .95	.99
	6	40	.25, .35, .40, .60, .65, .75	.58	.10, .75, .80, .85, .90, .95	.99
	8	30	.25, .30, .35, .40 .60, .65, .70, .75	.58	.05, .10, .70, .75 .80, .85, .90, .95	1.04
480	3	160	.30, .55, .65	.40	.05, .80, .85	.71
	5	96	.30, .40, .45, .65, .70	.39	.05, .80, .85, .90, .95	.68
	6	80	.30, .35, .40, .60, .65, .70	.39	.05, .75, .80, .85, .90, .95	.68
	8	60	.30, .35, .40, .45 .50, .60, .70, .75	.39	.05, .10, .70, .75 .80, .85, .90, .95	.72
720	3	240	.35, .50, .65	.32	.05, .80, .85	.57
	5	144	.30, .40, .55, .60, .65	.32	.05, .80, .85, .90, .95	55
	6	120	.30, .40, .45 .50, .65, .70	.32	.05, .75, .80, .85, .90, .95	.55
	8	90	.30, .35, .40, .45 .55, .60, .65, .70	.32	.05, .10, .70, .75 .80, .85, .90, .95	.58

[1]P_t = probability of response.

L* = DOSESCREEN measure of precision for optimal design.

well. *Table 3* presents designs that estimate the ED_{90} with progressively greater precision, at the expense of the precision of the ED_{50} estimate. With this tool, a compromise design can be chosen intelligently.

2.3 Determining Sample Size

Another possible application of DOSESCREEN is determining a sample size large enough so that the experiment will produce an ED confidence interval of a certain median length L_0. This application requires a prior estimate of β_2. If $\widetilde{\beta}_2$ is an estimate of β_2, DOSESCREEN can be put into a loop to determine N such that $L^* = L_0\widetilde{\beta}_2$, for fixed P_t and n_t/N, the proportional allocation of subjects to dose levels.

When using DOSESCREEN to determine sample size, L_0 must be specified in the same units as x_t. If, for example, x_t is in units of log-concentration—as is often the case in chemical bioassays—then L_0 must represent the desired length for a confidence interval for the log ED. This requirement limits the usefulness of DOSESCREEN for sample size determination because no simple relationship exists between confidence interval length for the log ED and the corresponding confidence interval length for the ED in the original units of concentration.

DESIGNING AN EXPERIMENT WITH DOSESCREEEN

To demonstrate how DOSESCREEN can be used to plan a chemical bioassay, we present a hypothetical experiment designed on the basis of DOSESCREEN output.

An experiment is being planned to investigate the toxicity of malathion to tent caterpillars. On the basis of a concentration-bracketing experiment, it appears reasonable to assume a probit model using a $\log_{10}$ transformation of the concentrations, which are in units of parts per million (ppm); intercept and slope estimates from the initial experiment were $\widetilde{\beta}_1 = 1.14$, $\widetilde{\beta}_2 = 1.01$. The planned experiment is primarily for the purpose of estimating the ED_{90}, but 95 percent confidence intervals for the ED_{50} and the ED_{95} will be reported as well. The investigator has 180 insects available for the study, and intends to use three to nine dose levels.

The following questions are explored by using DOSESCREEN interactively:

- For precise estimation of the ED_{90}, where on the interval (0,1) should the response probabilities lie?
- What is the effect of varying the number of dose levels?
- Should all doses have the same number of subjects, or can one improve precision by unequal allocation of subjects to doses?
- How well can the ED_{95} and ED_{50} be estimated with a design selected specifically to estimate the ED_{90}?

Using a design with five doses, a total sample of N = 180, and equal cell sizes, three possible distributions of response probabilities P_t, t =1,5, are considered: an even distribution between 0.10 and 0.90; placement of all the response probabilities on the upper half of the response curve; and an intermediate design, in which P_1 is on the low end of the response curve, the other P_t located on the upper portion.

P_t					L^* for ED_{90}
0.10	0.30	0.50	0.70	0.90	0.87
.50	.60	.70	.80	.90	1.09
.10	.60	.70	.80	.90	.71

On this evidence, a design that places most of the doses near the ED_{90} and a small number of P_t on the lower portion of the response curve is desirable.

The effect of varying the number of dose levels is next explored by considering the use of three, five, or nine levels, again with N = 180 and equal cell sizes:

P_t									L^* for ED_{90}
0.05	0.90	0.95							0.66
.05	.80	.85	0.90	0.95					.59
.05	.10	.15	.70	.75	0.80	0.85	0.90	0.95	.67

Five doses appear preferable to either three or nine doses.

There are numerous ways to allocate the 180 subjects to the five doses. Here, we consider three: equal allocation ($n_t = 36$), assignment of more subjects to P_t at the upper and lower extremes ($n_1 = n_5 = 54$, $n_2 = n_3 = n_4 = 24$), or assignment of more subjects to intermediate values of P_t ($n_1 = n_5 = 18$, $n_2 = n_3 = n_4 = 48$). When these allocation schemes are used with the best design encountered thus far, the following values of L^* are obtained:

		P_t			
0.05	0.80	0.85	0.90	0.95	L^* for ED_{90}
36	36	36	36	36	0.59
54	24	24	24	54	.63
18	48	48	48	18	.59

For estimation of the ED_{90}, two of the three allocation schemes appear to be equally precise. To choose between them, we examine their relative precision for estimating the ED_{95} and the ED_{50} at N = 180:

		P_t				
0.05	0.80	0.85	0.90	0.95	L^* for ED_{95}	L^* for ED_{50}
36	36	36	36	36	0.71	0.66
18	48	48	48	18	.78	.88

Clearly, the equal allocation results in higher precision for both the ED_{95} and ED_{50}, and is thus preferable to the allocation scheme that assigns a higher proportion of subjects to the central values of P_t.

At this point, we have narrowed the field of possible experimental designs to those with equal allocation of subjects to doses. Further, the doses will be placed so that one dose results in a low probability of response, with the others located so as to produce response probabilities between 80 and 95 percent.

The final step entails selection of doses that will faithfully reproduce the desired response probabilities. This is done by using the estimates of the regression coefficients and the identity

$$P_t = \Phi\left(\tilde{\beta}_1 + \tilde{\beta}_2 x_t\right)$$

or

$$x_t = \left[\Phi^{-1}(P_t) - \tilde{\beta}_1\right] / \tilde{\beta}_2$$

where $\Phi(\cdot)$ represents the standard normal cumulative distribution function and $\Phi^{-1}(\cdot)$ is its inverse. A computer program that computes $\Phi^{-1}(\cdot)$ or a standard normal table is required to solve for x_t. For our hypothetical experiment, the doses in the original units of concentration would be 10^{x_t}:

		P_t		
0.05	0.80	0.85	0.90	0.95
		ppm		
0.0017	0.51	0.79	1.39	3.13

4. APPENDIX

4.1 DOSESCREEN Subroutine and Auxiliary Subroutines

```
      SUBROUTINE DOSESC(Z,NDOSES,N,P,P0,W,FIP,LSTAR,IER)
C
C ******************************************************************************
C This subroutine computes LSTAR, a measure of precision for the estimation
C of an ED in a quantal response model with one independent variable.
C
C Argument list:
C      Input:  Z     : the (1-ALPHA/2)*100 percent quantile of the
C                       normal(0,1) distribution.
C
C              NDOSES: number of dose levels specified.
C
C              N(I)  : number of subjects at dose level I, I=1,NDOSES
C
C              P(I)  : probability of response at dose level I, I=1,NDOSES
C
C              P0    : the probability of response associated with the
C                       ED of interest. For example, if interest lies in
C                       the ED90, then P0 = 0.9.
C
C      Output: W(I)  : the weight associated with dose level I, I=1,DOSES
C
C              FIP(I): F inverse of P(I), computed by subroutine FUNCT,
C                       I=1,NDOSES
C
C              LSTAR : measure of precision.
C
C              IER   : error code
C                       IER = 1 if successful completion of computation
C                           = 2 if G is greater than or equal to 1.0,
C                               indicating that LSTAR is infinite.
C                           = 3 if input P(I) or P0 is not in (0,1).
C                           = 4 if N(I) is not greater than 0.
C                           = 5 if SIGMA or SIGMA inverse is nearly
C                               singular to within a tolerance of 1.0*E-8.
C
C
C subroutines called by DOSESC:
C
C      CHKPT:  verifies that P(I) is in (0,1) and that N(I) is at least 1.
C
C      FUNCT:  user supplied subroutine which computes F inverse of P0,
C              of P(I) and weights W(I) corresponding to user's choice
C              of model.
C
C ******************************************************************************
```

```
      DIMENSION P(NDOSES),N(NDOSES),W(NDOSES),FIP(NDOSES),SIGMA(2,2)
      REAL LSTAR
C
      CALL CHKPT(NDOSES,N,P,P0,IER)
      IF (IER.NE.1) RETURN
C
      CALL FUNCT(NDOSES,P,P0,FIP,FIP0,W)
C
C Compute SIGMA, the covariance matrix of the parameter estimates.
C
      SNW=0.
      SNWX=0.
      SNWXX=0.
C
      DO 100 I=1,NDOSES
              SNW=SNW + N(I)*W(I)
              SNWX = SNWX + N(I)*W(I)*FIP(I)
              SNWXX = SNWXX + N(I)*W(I)*FIP(I)*FIP(I)
 100  CONTINUE
C
C Check on singularity of SIGMA. Determinant of SIGMA is 1/DET.
C
      DET = SNW*SNWXX - SNWX*SNWX
      IF (DET.GT.1.E-8.AND.DET.LT.1.E8) GO TO 200
C
C Sigma will be nearly singular. Set IER to 5 and RETURN
C
      IER = 5
      RETURN
C
 200  SIGMA(1,1) = SNWXX/DET
      SIGMA(1,2) = -SNWX/DET
      SIGMA(2,2) = SNW/DET
C
C Compute G and verify that G is in interval (0,1)
C
      G=Z*Z*SIGMA(2,2)
      IF (G.GE.0.0.AND.G.LT.1.0) GO TO 300
C
C G is not in the interval (0,1).Set IER to 2 and RETURN.
C
      IER = 2
      RETURN
C
 300  A = 2.*Z/(1.-G)
      B = SIGMA(1,1) + 2.*SIGMA(1,2)*FIP0 + SIGMA(2,2)*FIP0*FIP0
      C = G*(SIGMA(1,1) - SIGMA(1,2)**2/SIGMA(2,2))
C
      LSTAR = A * SQRT(B-C)
      RETURN
      END
```

```
      SUBROUTINE CHKPT(NDOSES,N,P,P0,IER)
C
C ***************************************************************************
C This subroutine verifies that P(I) and P0 all lie in (0,1) and that
C N(I) is greater than or equal to 1, I=1,NDOSES
C
C ***************************************************************************
      DIMENSION N(NDOSES),P(NDOSES)
      IER = 1
C
C check on P0
C
      IF (P0.GT.0.0.AND.P0.LT.1.0) GO TO 100
C
C o.w. IER equals 3 and RETURN
C
      IER = 3
      RETURN
C
  100 DO 300 I=1,NDOSES
              IF (P(I).GT.0.0.AND.P(I).LT.1.0) GO TO 200
C
C o.w. IER equals 3 and RETURN
              IER = 3
              RETURN
C
  200         IF (N(I).GE.1) GO TO 300
C
C o.w. IER equals 4 and RETURN
C
              IER = 4
              RETURN
  300 CONTINUE
C
      RETURN
      END
```

```
      SUBROUTINE FUNCT(NDOSES,P,P0,FIP,FIP0,W)
C
C ********************************************************************
C This is a sample of a subroutine FUNCT which is appropriate when a
C Logit model is assumed; that is
C
C              P = F(X) = 1/(1 + EXP(-X))
C
C with density function
C
C              F'(X) = EXP(-X)/(1 + EXP(-X))**2
C
C                    = F(X)*(1-F(X))
C
C and inverse function
C
C              FI(P) = LOG(P/(1-P)).
C
C
C
C
C Therefore    FIP0 = LOG(P0/(1-P0)),
C
C              FIP(I) = LOG(P(I)/(1-P(I))), I=1,NDOSES
C and
C              W(I) = F'(FIP(I))**2/(P(I)*(1-P(I)))
C
C                   = P(I)*(1-P(I)), I=1,NDOSES.
C
C ********************************************************************
      DIMENSION P(NDOSES),FIP(NDOSES),W(NDOSES)
C
      FIP0 = ALOG(P0/(1.-P0))
C
      DO 100 I=1,NDOSES
              FIP(I) = ALOG(P(I)/(1.-P(I)))
              W(I) = P(I)*(1.-P(I))
  100 CONTINUE
C
      RETURN
      END
```

4.2 Sample Main Program and Output

```
C ****************************************************************************
C This is a sample main routine which calls the DOSESC subroutine. In this
C example, LSTAR is computed for the ED50; three dose levels are proposed,
C with 50 subjects assigned to each dose level. Response probabilities
C are 0.20,0.50, and 0.80.
c
C The value of Z associated with ALPHA = 0.05 is input directly here.
C To increase the accuracy of Z, one may supply a subroutine that
C numerically evaluates the inverse gaussian distribution function at
C the value (1-ALPHA/2).
C
C **************************************************************************
      DIMENSION P(3),N(3),W(3),FIP(3)
      REAL LSTAR
      DATA ALPHA/.05/,Z/1.95996/,N/3*50/,P/.2,.5,.8/,P0/.5/,NDOSES/3/,
     1 NIO/6/
C
      CALL DOSESC(Z,NDOSES,N,P,P0,W,FIP,LSTAR,IER)
C
      WRITE (NIO,20)
      DO 10  I=1,NDOSES
              WRITE(NIO,30) N(I), P(I)
 10   CONTINUE
 20   FORMAT (10X,"DOSESCREEN ESTIMATE LSTAR"/
     1 "             N              P          ")
 30   FORMAT (10X,I4,10X,F4.3)
C
      GO TO (100,200,300,400,500), IER
C
 100  NP0 = 100*P0
      NSIGN = 100*(1.-ALPHA)
      WRITE(NIO,101) LSTAR,NSIGN,NP0
      STOP
 101  FORMAT (10X,"LSTAR = ",F10.4," FOR A ",I2,"% CI FOR THE ED",i2)
C
 200  WRITE(NIO,201)
      STOP
 201  FORMAT(10X,"ESTIMATED CONFIDENCE INTERVAL FOR THIS DATA INFINITE")
C
 300  WRITE(NIO,301) P0
      STOP
 301  FORMAT(10X,"P(I) OR P0 = ",F7.3," NOT IN THE INTERVAL (0,1)")
C
 400  WRITE(NIO,401)
      STOP
 401  FORMAT(10X,"N(I) NOT AN INTEGER GREATER THAN OR EQUAL TO 1")
C
 500  WRITE(NIO,501)
      STOP
 501  FORMAT(10X,"AN ALMOST SINGULAR COVARIANCE MATRIX DETECTED")
C
      END
```

Output from sample Main program

```
DOSESCREEN ESTIMATE LSTAR
  N          P
  50        .200
  50        .500
  50        .800
LSTAR =    .0641 FOR A 95% CI FOR THE ED50
```

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Careful selection of an experimental design for a bioassay substantially improves the precision of effective dose (ED) estimates. Design considerations typically include determination of sample size, dose selection, and allocation of subjects to doses. DOSESCREEN is a computer program written to help investigators select an efficient design for the estimation of an arbitrary ED. This report establishes the statistical basis for DOSESCREEN and suggests several ways to utilize DOSESCREEN output. A copy of the computer program in Fortran 77 is provided so users can easily convert DOSESCREEN to their own computer system.

Retrieval terms: bioassay, experimental design, effective doses

United States
Department of
Agriculture

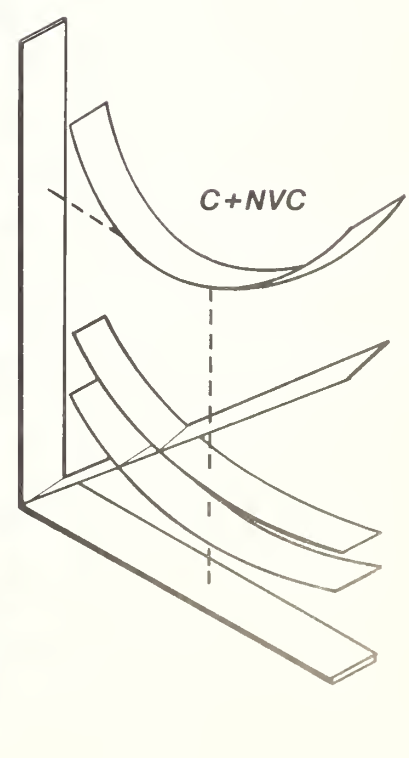
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-79

Methods for Assessing the Impact of Fire on Forest Recreation

Henry J. Vaux, Jr. **Philip D. Gardner** **Thomas J. Mills**



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Acknowledgments:

We thank Robert W. Mutch of the Lolo National Forest for his helpful technical advice, and Stewart Fox and Katherine Stenberg for their able research assistance while they were graduate students in environmental administration, University of California, Riverside, California.

Cover: Recreationists were shown photographs taken of a recreation site before (top) and after (middle and bottom) a wildfire, and asked about their willingness to pay an entrance fee to the site.

Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

September 1984

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IN BRIEF

Vaux, Henry J., Jr.; Gardner, Philip D.; Mills, Thomas J. **Methods for assessing the impact of fire on forest recreation**. Gen. Tech. Rep. PSW-79. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1984. 13 p.

Retrieval terms: recreation economics, travel cost demand models, contingent market valuation, wildfire impacts, landscape attributes, wildland fire management

Are methods for measuring outdoor recreation demands sensitive to quality differences caused by fire in forest environments? And what methods are the most suitable for estimating recreation demand and values in fire-affected forest situations? A literature search and an experiment were conducted to answer those questions.

Both economic and psychological methods could be used to evaluate the effects of fire on forest recreation. These methods rely on direct and inferential means to assess the values of outdoor recreation. The most suitable of these approaches appears to be contingent market valuation—a direct, economic technique that uses personal interviews. A hypothetical market transaction environment is set up within which values are estimated.

This approach has been used to assess the impact of insect infestations and of timber cutting on forest environments. The effects of such infestations and cuttings are similar to the effect of fire. Recreationists' preferences for visual qualities of the landscape are measured by having them rate photographs according to relative attractiveness. These photographs vary from no insect damage or no cutting to changes in scenic attributes caused by these disturbances. This approach seemed appropriate to the study of the impact of fire on outdoor recreation values.

An illustrative application of this contingent market valuation approach was undertaken with 69 students at the University of California, Riverside. They were shown pairs of photographs identified as typical National Forest sites available for recreational activites. Each person was asked to express a preference among the paired photographs. These pictures depicted scenes before and after a fire. In addition, the respondents were asked about their willingness-to-pay an entrance fee to a recreation site by an iterative bidding procedure. A mathematical equation was developed to compute the present discounted value of the recreation activity in a fire-free site and in a fire-affected site, and to estimate the respondents' willingness-to-pay.

The illustrative results suggest that less intense fires may have beneficial economic effects, whereas intense fires may have detrimental effects on recreation values. The valuation of the impact of fire was not always negative or unanimous among respondents, and preferences are subject to change. Willingness-to-pay is an appropriate measure for valuing the effects of fire on forest recreation.

Efforts to evaluate the effects of fire on natural resources have focused largely on impacts that are commonly measured in monetary terms, such as losses of commercial timber, grazing yields, and structures. The effects on the generally unmarketed resources, such as watershed, fish, wildlife, and recreation, have been considered in policy directives, but the economic implications of the effects of fire on these forest amenities have not been fully understood. These latter resource categories have figured less prominently in fire management program evaluations partly because they are not usually traded in markets.

Recreation is one resource that is least subject to market allocation, and the economic literature on outdoor recreation presents a varied and sometimes conflicting array of conclusions. Investigators have reached no consensus on the magnitude of recreation benefits or the appropriate methodologies for estimating these benefits. Successful attempts to isolate the value of individual attributes of the outdoor recreation experience have been few.

Efforts to assess the impact of fire on recreation usage and value are even more sparse. The fact that fire affects only selected site attributes, and also affects the path of those attributes over time further confounds the measurement problem (Davis and others 1980). The measurement problems can be diagramed: Initial State → State A → State B → State C. Given some initial state and the absence of fire, a specific habitat will evolve along a successional continuum to subsequent states, labelled A, B, and C for simplicity. If a fire occurs at the initial state, a different successional trajectory will follow with States A′, B′, and C′: Initial State → State A′ → State B′ → State C′.

This report assesses the degree to which current methods for measuring recreation demand are sensitive to quality differences caused by fire in different forest environments, and recommends methods for estimating recreation demands and values in a fire-affected situation.

VALUING OUTDOOR RECREATION

Work on valuing unmarketed commodities has led to two distinct approaches. Direct techniques rely on survey instruments and interviews to query recreationists directly about the values they ascribe to recreation. Inferential techniques infer from observed behavior the willingness of a person to pay for outdoor recreation activities. Both approaches are based on the theory that the total value of the commodity is measured by the

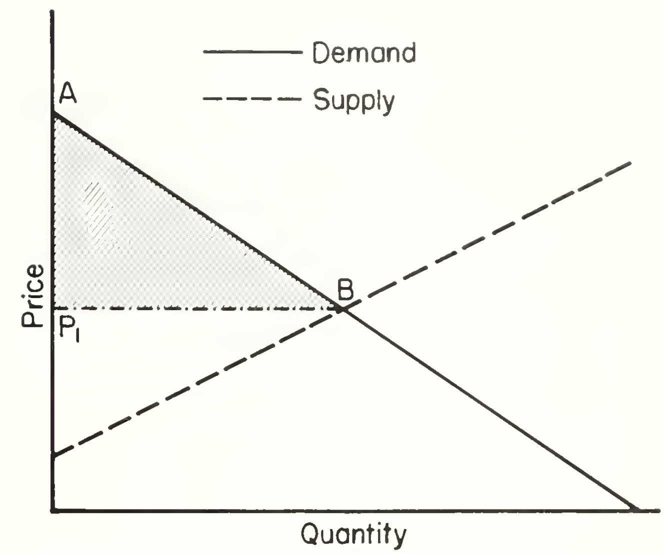


Figure 1—Consumer surplus—what a consumer is willing to pay beyond what is actually paid—is represented by the shaded area, with the prevailing price of a marketed good shown as P_1.

consumer's willingness-to-pay for it. Consumer surplus is the amount a consumer would be willing to pay for a commodity *in excess* of what he actually pays (Mishan 1959). Graphically, consumer surplus is approximated as the area under the Marshallian demand curve in excess of the prevailing price (*fig. 1*). The shaded area ABP_1 is the total consumer surplus when the prevailing price of a marketed good is P_1 (*fig. 1*). Where the good is not priced because of the absence or failure of markets, the consumer surplus is the entire shaded area under the Marshallian demand curve, ACD (*fig. 2*). The consumer surplus for unpriced goods is equivalent to total willingness-to-pay.

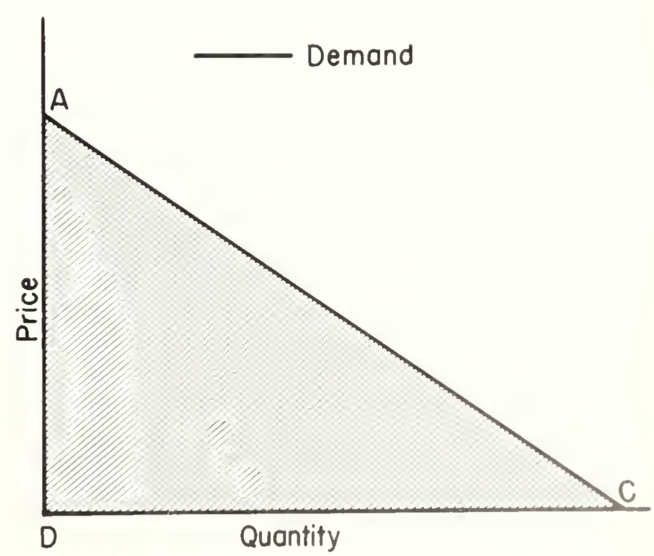


Figure 2—The shaded area under the demand curve represents consumer surplus where no market price exists—what the consumer is willing to pay beyond what is actually paid.

Inferential Techniques

The premise behind inferential techniques is that willingness-to-pay can be inferred from the observed consumer behavior. In general, inferential models are useful where the commodity or its attributes can be specified quite explicitly. Three classes of inferential models exist: (a) travel-cost demand, (b) consumer theory, and (c) gravity. We focused on travel cost demand models because they have been used more extensively in recreation studies than the other two classes.

In implementation of the travel cost demand method, the analyst first specifies a number of travel zones from a recreation site (see for example, Cicchetti and others 1972, Clawson 1958, Hotelling 1938, and Pearse 1968). The residence of each recreationist is then used to estimate his or her participation rates for each travel zone. The inverse relationship between participation rates and distance from the recreation site is the basis for constructing the demand curve. Recreationists from the furthest travel zone are assumed to be the marginal consumers. The consumer surplus of the site is estimated as the sum of the rents per unit of activity for all points of origin multiplied by the number of trips from each origin. All other consumers are assumed to react to an increased entrance fee to a recreation site in the same way that they would react to an increase in travel costs.

The typical travel cost demand model assumes that (a) tastes and preferences are identical both within and between different distance zones, (b) recreationists react to changes in fees in the same manner as to changes in the travel costs, (c) alternative recreational opportunities are equally available for all populations, (d) travel time is neglected and all consumers have the same opportunity cost for time, (e) the population has homogeneous social and economic characteristics, and (f) all visitors spend the same amount of time at the recreation site.

The travel cost assumptions about homogeneity of tastes, preferences, and alternatives have been refined by analyzing only people who actually participate in recreation and by stratifying them by income class (Pearse 1968). Further substratification within income classes by occupation, age, or other socio-economic variables was suggested as further refinement. The assumptions are still arbitrary, however, and willingness-to-pay varies inversely with the number of arbitrary income classes.

Recreationists react to a host of variables that determines visitation rates—not simply to changes in travel costs alone—and the value of time for travel must also be recognized (Knetsch 1963). Visitation rates may not fall as much in response to fee increases as predicted by the travel costs method (Cesario and Knetsch 1970, 1976). The principal source of bias may be the neglect of travel time and not the failure to account for the total time spent in recreation. The opportunity cost of time spent in recreation cannot, however, be ignored (McConnell 1975, 1976). A study of these conflicting views lead to the conclusion that as a general principle, on-site time costs should be included (Wilman 1980).

The need to include differing qualities of alternative recreation sites has been addressed in several studies. An empirical method to distinguish the quality value of a site from its locational attractiveness proposes that a site quality difference exists when (a) visitation rates to various sites are unequal, or (b) visitation rates are equal although travel costs are unequal (Wennergren and Fullerton 1972). The impacts of individual quality attributes cannot normally be estimated with this method, however, because of the inability to pair sites which are similar in all attributes other than the one under study.

Distortions can be introduced into travel costs demand analysis through data aggregation. The aggregation of data from visitor records usually used to calculate the demand curve of the average individual interjects distortions (Smith 1981). All individuals are represented by the same demand function even though they may engage in different activities during a recreational visit and vary the timing of their visits. The masking of these distinctions through data aggregation could lead to faulty estimates of recreation benefits.

Travel cost demand methods have two distinct advantages. First, they rely on observations of actual behavior by recreationists. This practice is consistent with that of using revealed preference methods common to demand analysis of marketed goods. Second, much of the data used are routinely gathered.

Offsetting these advantages of travel cost models are disadvantages which severely limit their potential use for fire impact analysis. These models are not well suited for estimating the value of individual quality attributes or for estimating changes in value over time. An aspect of the travel cost demand methodology that is a problem is its assignment of values to the entire recreation experience. Wennergren and Fullerton (1972) made some advances in the separation of the value contribution of location and quality characteristics, but they did not provide an *a priori* means for distinguishing between the components of value attributable to specific qualities. The impact of fire affects only selected quality attributes.

The travel cost demand method could be used to estimate the value of specific characteristics only if pairs of sites could be found that were identical in location and quality except for the attribute under investigation, e.g., sites homogeneous in all respects except for fire histories. Alternatively, heterogeneous sites could be used if there were some way to standardize for all differing quality attributes. Either of these solutions to the method's deficiences would, however, increase the costs of data acquisition enormously.

The impacts of fire on the landscape are not static, and inferential methods are especially ill-suited to account for changes over time. Actual recreation behavior over time that would be measured by inferential methods is influenced largely by changing economic and cultural factors as opposed to variation in the physical landscape where recreation occurs (Clawson and Knetsch 1966). In short, because of the problems of measuring the impact of individual site attributes, especially when those attributes change over time, inferential methods do not hold much promise for assessing the impact of fire on recreation.

Direct Techniques

Direct methods of determining consumer willingness-to-pay are characterized by personal interviewing techniques, such as contingent market valuation approaches. In such approaches, a hypothetical market environment is developed within which values are estimated. An interview can elicit the highest possible bid from individual recreationists for the realistic, albeit hypothetical, recreational service or product (Davis 1963). That bid represents the potential user's maximum willingness-to-pay for that service or product. Alternatively, the bid can be considered the minimum compensation required to induce the user to forego the service or product.

Direct questioning techniques have been criticized on two points (Samuelson 1954). Individuals may have incentives to give false bidding signals, i.e., follow some bidding strategy that does not reflect their true willingness-to-pay, dependent upon how they believe their bids may be used in setting fees and depending on the perceived role of the interviewer. Bids given under a hypothetical situation may not be the same as those that would be paid in the real-life situation.

To test for the presence of strategic bidding, Bohm (1971, 1972), designed an experiment to make the choices so complicated that possibilities for strategic behavior would be obscured and consumers would thus state their true maximum willingness-to-pay. Respondents were told that (a) if aggregate bids were too low, the product in question will not be produced; (b) overstated bids would increase the probability that the consumers would have to pay the groups average bid; (c) it was morally wrong to give false bids; and (d) there was no anonymity in the final bids. In other experiments, testing for strategic behavior and hypothetical bias was done by employing different sets of instructions to bidders and by emphasizing five possible bidding-to-payment outcomes (Bohm 1972). The results suggested that hypothetical questions containing a minimum of information tended to induce hypothetical behavior.

The sensitivity of alternative payment vehicles with realistic, if hypothetical, situations related to air quality degradation was tested by the use of photographs (Randall and others 1974). Payment vehicles included (a) sales tax rate increases, (b) added charges to their electricity bills, or (c) increases in park user fees. The bids appeared to be sensitive to the payment vehicle.

In a study of air quality degradation in a scenic area, a single bidding technique was used to test for strategic behavior by postulating that honest bids would be normally distributed (Brookshire and others 1976). The results were roughly consistent with those reported by Randall and others (1974).

The value of visibility reductions was investigated in the Farmington (New Mexico) area in a systematic test for potential strategic, information, and hypothetical biases (Rowe and others 1980). Prior to bidding, some respondents were provided with hypothetical information about the mean bids of other respondents. Those not supplied with this information were told after the bidding and asked if they would change their bid. The findings showed no significant strategic bias. Visibility reduction was also examined in a study of geothermal energy development in the Jemez Mountain region of New Mexico by using a series of photographs and a questionnaire focused on site substitution possibilities (Thayer 1981). The results showed that neither information nor starting point bias was statistically significant. Hypothetical bias was analyzed by comparing the bids and site substitution preferences with estimates of the additional travel costs to substitute sites. An analysis of the incremental cost of travel associated with substitutes provided a useful means to cross-check the validity of bids (Thayer 1981).

Elk hunters in the Laramie (Wyoming) area were surveyed to determine the values which hunters placed on the right to hunt elk, the differing habitats in which elk hunting occurred, and the number of daily encounters with elk (Brookshire and others 1980). No starting point bias was found but bids appeared to be influenced by the payment vehicle. The respondents reacted negatively to the utility bill vehicle, but not to the hunting license fee vehicle.

The results of a contingent market valuation study of air quality impacts were compared with those of a study of real market housing prices (Brookshire and others 1982). The differences in housing values compared quite favorably with the willingness-to-pay bids although the bids were somewhat lower. The bids were sensitive to housing characteristics, levels of income, knowledge of air pollution effects on health, and other characteristics studied.

Although contingent market valuation studies have been viewed with skepticism, a growing literature suggests that much of the skepticism is unwarranted. A review of the major contingent market studies found the results surprisingly consistent (Schulze and others 1981). Strategic and information biases did not appear in most instances where efforts were made to test for their existence. The principal problem remaining is with hypothetical bias and the inability to test for it. Even with hypothetical bias, however, contingent market methods can estimate values of nonmarketed goods that cannot be estimated in any other way.

Contingent markets, even though fictional, generate estimates more consistent with neoclassical consumer theory than do inferential methods (Brookshire and Crocker 1981). Contingent market methods measure the value consumers expect to glean from a nonmarketed commodity, not the value ultimately realized. Inferential methodologies fail to fully account for consumer expectations by estimating the value a consumer actually obtains. Contingent market methods also simplify the problems of accounting for the effects of extraneous and confounding variables. In the absence of more detailed analytical study of how expectations are formed, however, it is not possible to confirm or refute the empirical findings from contingent markets. On the balance, contingent valuation methods appear best suited for evaluating the impact of fire on forest recreation experiences.

METHODS FOR MEASURING IMPACT OF FIRE

Most recreation studies have focused on the quantitative values of the entire recreational experience rather than on the effects of individual forest environment attributes. Two types of studies that measure the value of qualitative changes in forest environments attributable to insect infestations and timber cutting practices are (a) economic studies that measure consumer surplus directly and provide explicit measures of the value of attributes; and (b) psychological studies that focus on the perceptions of recreationists *per se* and do not derive direct measures of economic value.

Economic Tests

Economic studies focus almost exclusively on forest insect infestations, but the effects of such infestations may be similar to the effects of fire. A study of a mountain pine beetle infestation on recreational sites in Idaho used a conventional travel-cost demand model (Michalson 1975). Demand functions for areas with and without pine beetle damage were derived from a questionnaire. The estimated value of each visitor-day (consumer surplus) range from $15.50 for the infested sites to $17.90 for the noninfested sites—a difference attributed entirely to insect infestation. A fundamental problem with that study is that differences in consumer surplus are attributed to a single characteristic of site quality. Such differences can only be attributed to a single quality when all of the other qualities are similar between sites.

A travel-cost demand model was combined with a gravity potential model to estimate the loss in recreation value at campgrounds infested with southern pine beetle in Texas (Leuschner and Young 1978). The major source of data were annual visitation records. Estimated annual damages ranged from $3,500 at sites with a 10 percent level of infestation to $700,000 at sites with a 90 percent level of infestation. Failure to consider substitution possibilities resulted in systematic overstatement of damages.

The economic impact of a gypsy moth infestation was examined in the northeastern United States (Moeller and others 1977). Data from personal interviews showed that all types of property owners were sensitive to infestation by the gypsy moth, though for different reasons. Recreational activity declined by 12 to 52 percent, depending upon the extent of damage and the type of operation.

The shortcomings of these travel-cost demand studies underscore the need to select sites with similar quality attributes, and the difficulty of accounting for substitution possibilities. The work also underscores the significance of the dynamic effects of disturbances in forest environments.

Psychological Tests

Social scientists have developed methods of assessing individual's preferences for the visual qualities of landscapes that focus on the direct specification of utility rather than measurement of consumer surplus (Daniel and Boster 1976, Kaplan and others 1972, Zube 1973). Several of these psychological utility methods have been used to assess the impact of insect infestations and timber cutting practices in forest environments.

Photographs of mountain pine beetle infestation in Idaho were used to assess the impact of insect damage on recreationist's preferences (White 1977). The photographs were selected to standardize for the distance or perspective of view. Recreationists rated the photographs according to the relative attractiveness. The preference analysis revealed that the distance or perspective significantly affected preferences. Close perspectives were preferred over more distant ones, and recreationists reacted more intensely to infestation damage in the near perspectives. The detrimental impact of insect damage was deemed to be offset by the presence of other scenic attributes in the landscape.

In a study of scenic preference functions for forested sites with varying amounts of southern pine beetle damage, training in forestry had no significant effect on preferences (Buhyoff and Leuschner 1978). Preferences for undamaged stands increased sharply as damage levels approached 10 percent, but preferences changed very little to damage levels beyond 10 percent. The inconsistencies between this study and the one by White (1977) show that the research method and the structure of the sample can influence the results.

The attractiveness of forest scenes was measured by focusing on the spatial quality of stands (Brush 1979). The results showed that participants had strong preferences for open stands irrespective of the species composition. This underscores the fact that disturbance phenomena that alter the density of stands over time may lead to evolution of more attractive stands even though the initial visual impact is not attractive. Certain fire-caused tree mortality has the possibility of creating such a stand structure.

The impact of a Douglas-fir tussock moth outbreak on recreation business was analyzed in northeastern Oregon (Downing and others 1977). The majority of interviewees cited factors other than the infestation as being critical determinants of the magnitude of recreation activity.

The more prominent study of preferences for forest attributes focused on the visual impacts of cutting practices in central Michigan (Langenau and others 1977). A series of sites was prepared with cutting controlled at 25, 50, and 75 percent levels. Two sites were prepared at each cutting intensity to control for quality attributes not related to cutting. An "untreated" site cut at a 3.5 percent intensity level was also prepared. Perceptions of individuals residing near the prepared sites were studied in 1974 and 1976 to test the hypothesis that individuals would tend to become "conditioned" to clearcutting, especially as the sites regenerated. The attitudes of those

exposed to the cutover sites did not change significantly over the 2-year period, possibly because experience had not altered prestudy attitudes about clearcutting practices. Or perhaps the number of encounters with cutting areas did not increase significantly between the first and second survey.

The relationship between the preferences for cutover landscapes and the diversity of recreation activities on the sites was studied by using the same study area (Levine and Langenau 1974). A recreation activity profile study showed that respondents with high diversity patterns of recreational activity were positively correlated with support for clearcutting practices. Recreationists who engage in a variety of activities were more likely to perceive the positive features of clearcutting than those who engage in only one activity. The intensity of recreation activities, as measured by the number of recreation trips was not related to support for clearcutting.

The impact of clearcut size and the extent of regeneration were related to the participation rates of recreationists and their perceived quality of the recreation experience (Langenau and others 1980). Attitudes toward clearcutting were strongly dependent upon the type of recreation activity. For example, campers were more concerned with the immediate surroundings while others focused on the totality of the landscapes. Relationship of recreational use and perceived site quality to cutting level suggests that the regenerative features of vegetative succession may be an important determinant of perceptions.

MEASURING RECREATION VALUES

Evaluating the impact of wildfire on forest recreation differs in an important respect from valuing other qualitative impacts. The effects of fire depend heavily on time—much more so than most other effects, except impacts from insect infestations and timber harvesting.

Fire can alter the successional path or trajectory of ecosytems over time and the qualitative effects of fire on the landscape may persist over long periods. The fire-free landscape evolves also. Therefore, a recreationist's perception of relative attractiveness of a fire-impacted and nonfire-impacted landscape depends on how much time has elapsed since the fire occurred and the differences in the vegetative succession of the two landscapes.

Depicting Fire Succession

The impact of fire varies with many factors, including the successional stage of the habitat and the intensity of the fire (Davis and others 1980). Any effort to value the impact of fire must be carried out within the habitat and successional context in which the fire occurs.

Contingent market valuation methods using iterative bidding procedures with photographs of similar environments, both with and without the effects of fire, can provide a reasonably straightforward means of standardizing the physical landscape. In fact, photographs appear to be the only practical way to control the factors unrelated to fire.

The photograph sequences can be obtained in two ways. One is to obtain pictures of the same site over time, from a period before the fire through points in the postfire regeneration. When photographs of the initial state are paired with each of the succeeding states, they represent various time points on the successional paths of the fire-impacted and fire-free sites. This is valid if the prefire scene is in a vegetative climax state that would remain unchanged in the absence of fire.

An example of this approach is a photograph sequence in a climax setting in Neal Canyon, central Idaho. The site was initially a Douglas-fir climax habitat type and the scene of an intense fire. The initial, or prefire state, photograph shows a climax state whose appearance over substantial periods of time can be represented by a single photograph (*fig. 3A*). The remaining photographs were taken from the same point 1 day after (*fig. 3B*), 3 years after (*fig. 3C*), 6 years (*fig. 3D*), and 12 years (*fig. 3E*) after the fire. A similar sequence was photographed from a distant perspective: the prefire site, (*fig. 4A*); and 1 days (*fig. 4B*) and 12 years (*fig. 4C*) after the fire.

If the initial state is not in a climax state, it is possible to construct, with the aid of fire ecology experts, sequences of pictures which provide close facsimiles of vegetation successions with and without fire. Care must be undertaken to ensure that extraneous influences are absent. The scarcity of sequence photos together with difficulties in constucting perfectly matched sequences suggest that it may not always be possible to eliminate extraneous influences such as photo orientation and amount of sunlight.

An example of a constructed sequence shows the effect of light ground fire on a ponderosa pine type habitat in the Sierra Nevada of California. They showed a successional trajectory in the absence of fire (*figs. 5A-5D*) and in its presence (*figs. 5E-5H*). These sequences yielded four pairs of photographs depicting the state of the trajectories after identical lapses in time from the initial state.

The three sequences in *figures 3-5* depict scenes that had no especially unique qualities. And so we included a simple sequence of fire in the presence of outstanding scenic attributes, the Garden Wall area of Glacier National Park, Montana—as it was in 1957 (*fig. 6A*), and in 1980 (*fig. 6B*). A major fire occurred in the Garden Wall area in 1967.

Most previous contingent market valuation studies measure only the intensity of preferences, under the implied assumption that the attribute is unanimously desirable or undesirable. There may not be unanimity, or even near unanimity, over which successional state is preferred. It is also possible that different successional paths will be preferred at different stages. A contingent market valuation study must be structured to identify this lack of unanimity and preference shifting.

Evaluating Preferences

To examine this preference structure, we conducted a simple preference rating test among 69 voluntary graduate and undergraduate students at the University of California, Riverside. Each participant was shown pairs of photographs identified as typical National Forest sites available for recreation activities and was asked to choose the preferred site. Each respondent rated the paired pictures from four sequences of pictures shown in *figures 3, 4, 5,* and 6.

The process of eliciting the respondents' preference and bidding was as follows. The interviewer would say:

> These photographs show two similar forest environments. One of the environments has been affected by fire and the other has not. Let's assume that these environments are generally representative of places where you can spend most of your time here in the forest. Do you prefer one to the other? ______________ A
> ______________ B

After the respondent had expressed a preference, we asked:

> Now, let's assume that by paying an entrance fee for your family or group you can be admitted to the recreation site that you prefer. This will be the only way to guarantee that you have access to such sites. Let's also assume that all visitors to the site will pay the same daily group or family fee as you and that all money collected will be used to finance fire control programs designed to preserve and maintain such sites. Would you be willing to pay $1.00 per day family fee to ensure that you could visit the site that you prefer? $2.00 per day?

We used an iterative bidding process in which the original bid was changed by increments of $1.00 per day until a negative response was obtained. Then we decreased the bid by 25¢ per day until a positive response was obtained, and recorded the amount.

The differences in the paired photos were explicitly attributed to fire and a connection was made with recreation activities. This characterized the "good" to be traded in the contingent market as precisely as possible. The payment vehicle—an entrance fee—was explicitly identified and the implications of that payment vehicle for others were also identified. Previous work had suggested that direct payment vehicles yield the most consistent results and are less prone to the bias of extraneous issues related to the payment method. The bidding sequence could be changed to incorporate different starting points and bid increments or decrements to test the sensitivity to the essentially arbitrary levels. And, additional questions could be added to test explicitly arbitrary levels. And, additional questions could be added to test explicitly for various types of strategic and information bias.

A majority of the respondents preferred the fire-free successional trajectory for the high-intensity fire site (*fig. 3*), but the preference was not unanimous (*table 1*). The decline in percentage preference for that sequence over time suggests that preference may decay further as the direct fire scars heal over time.

Table 1—*Preferences for fire-impacted and fire-free landscapes for the intense fire sequence from* figures 3 *and* 4

Picture pair	Time	Ordinal preferences: Burned scene	Ordinal preferences: Unburned scene
		Percent	
	Intense fire sequence—near perspective		
Figure:			
3A, B	1 day	11.4	88.6
3A, C	3 years	14.3	85.7
3A, D	6 years	22.9	77.1
3A, E	12 years	30.9	70.0
	Intense fire sequence—far perspective		
Figure:			
4A, B	1 day	8.5	91.5
4A, C	12 years	8.7	91.3

The fire-free trajectory for the low-intensity fire site *(fig. 5)* was preferred by a majority of the respondents only in the immediate postfire period. Thereafter, the fire-impacted trajectory was preferred by a majority, especially 5 years after the fire and beyond. This preference switch demonstrates the need for a methodology which captures preference ratings at several points on the successional paths (*table 2*). Those results suggest that less intense fires may have beneficial economic effects on outdoor recreation, whereas intense fires may have detrimental effects.

The lack of unanimous preferences raised the further issue of how different preferences ought to be weighted. If forest resources are to be allocated efficiently, it is appropriate to weight preferences according to recreationists willingness-to-pay.

The problem of measuring preference can be characterized more formally to calculate the present discounted value of recreational activity in the fire-free trajectory and the fire-affected trajectory by the equations:

$$V_1 = \int_0^t WTP_1(t)e^{-rt}dt. \tag{1}$$

$$V_2 = \int_0^t WTP_2(t)e^{-rt}dt \tag{2}$$

Table 2—*Preferences for fire-impacted and fire-free landscapes for the low intensity fire sequence from* figure 5

Picture pair (figure)	Time	Ordinal preferences: Burned scene	Ordinal preferences: Unburned scene
		Percent	
5A, E	Initial	20.3	79.7
5B, F	6 months	56.5	43.5
5E, H	5 years	89.9	10.1
5D, H	10 years	87.1	12.9


A


B


C


D


E

Figure 3—Changes in vegetative succession are depicted in this near perspective of a Douglas-fir climax site in Neal Canyon, central Idaho, before fire *(A)* and at varying times—1 day *(B)*, 3 years *(C)*, 6 years *(D)*, and 12 years *(E)*—after fire.

A

B

C

Figure 4—Changes in vegetative succession are depicted in this far perspective of a Douglas-fir climax site in Neal Canyon, central Idaho, before fire *(A)* and 1 day *(B)* and 12 years *(C)* after fire.

A

Figure 5—Changes in vegetative succession are depicted in a typical ponderosa pine type habitat, Sierra Nevada of California, that had *not* been affected by fire—the site when it was initially photographed *(A)*, and 6 months *(B)*, 5 years *(C)*, and 10 years *(D)* later. A similar habitat but one affected by fire is shown at the time it was initially photographed *(E)*, and 6 months *(F)*, 4 years *(G)*, and 10 years *(H)* later.

G

B

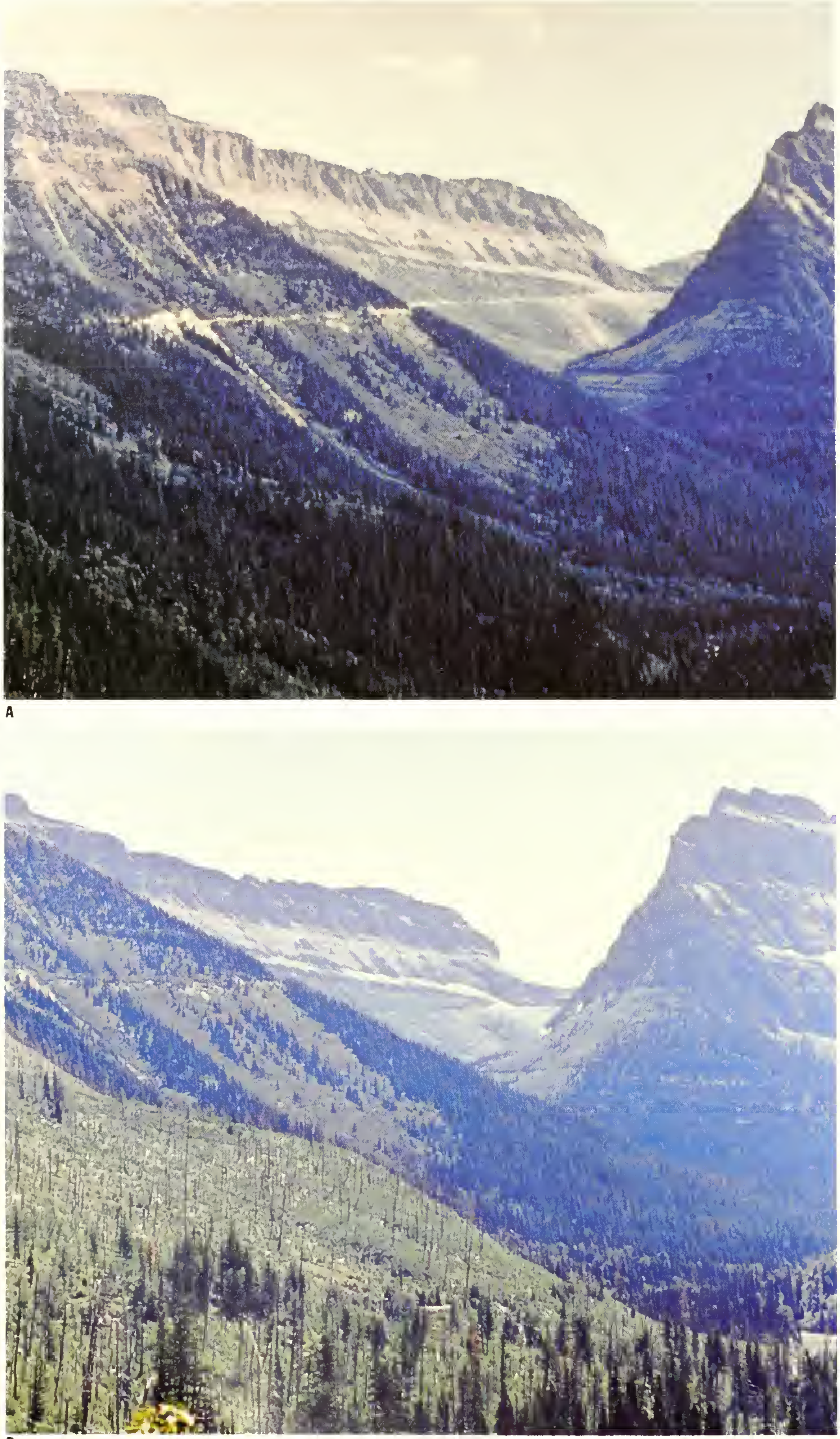
C

D

E

F

H



A

B

Figure 6—Scenic attributes in the Garden Wall area, Glacier National Park, Montana, have changed during the period from 1957 *(A)* to 1980 *(B)*. A major fire occurred in the area in 1967.

in which

V_1 = total present value of recreation activity in the fire free successional trajectory

V_2 = total present value of recreational activity in the fire affected successional trajectory

$WTP_1(t)$ = willingness-to-pay at time t for fire free trajectory

$WTP_2(t)$ = willingness-to-pay at time t for fire affected trajectory

r = discount rate

t = period of analysis

The net value impact of the fire can be expressed by subtracting V_2 from V_1. The result may be either positive, indicating a net "loss" in value, or negative, indicating a net "gain" in value, depending upon which trajectory is preferred.

Careful selection of the successional trajectory time intervals between preference measurements permit the willingness-to-pay functions to be approximated by linear segments within each time interval. The specification of time intervals will crucially affect the data requirements, however. If the discrete intervals are defined as $t_1 + t_2 + t_3, \ldots, t_n$ (in which $t_1 + t_2 + t_3 \ldots + t_n = T$), the additivity property permits equations 1 and 2 to be rewritten as follows:

$$V_1 = \int_0^{t_1} WTP_1(t)e^{-rt}dt + \int_{t_1}^{t_2} WTP_1(t)e^{-rt}dt + \int_{t_2}^{t_3} WTP_1(t)e^{-rt}dt + \ldots + \int_{t_{n-1}}^{t_n} WTP_1(t)e^{-rt} \quad (3)$$

$$V_2 = \int_{t_0}^{t_1} WTP_2(t)e^{-rt}dt + \int_{t_1}^{t_2} WTP_2(t)e^{-rt}dt + \int_{t_2}^{t_3} WTP_2(t)e^{-rt}dt + \ldots + \int_{t_{n-1}}^{t_n} WTP_2(t)e^{-rt}dt \quad (4)$$

The computation of willingness-to-pay requires an estimate of the total willingness-to-pay at the upper and lower bounds of integration and the appropriate interest rate. Total willingness-to-pay at any time, t, and for any trajectory, j, can be estimated by multiplying the average (sample) individual willingness-to-pay by the estimated number of recreationists in the total population that prefer that trajectory. The individual's (sample) willingness-to-pay can be obtained for a sample population, such as the study using 69 students.

The sample willingness-to-pay must be combined with estimates of the visitation rate for the site in question to obtain total willingness-to-pay as indicated in expression (5). The visitation rates must be estimated separately and should account for substitution possibilities. In forest settings where there are many substitution possibilities, it will usually be safe to assume that visitors can substitute away from less preferred sites at little, if any, additional cost. In those instances, aggregate willingness-to-pay for the preferred trajectory may be quite small. If the expense of fire damage is large or the site in question is unique, substitution will be more costly and aggregate willingness-to-pay will be correspondingly higher.

$$WTP_{jt} = \frac{\sum_{i=o}^{n_{jt}} WTP_{ijt}}{n_{jt}} \cdot N_{kt} \quad (5)$$

in which

WTP_{jt} = total population willingness-to-pay for the jth trajectory at time t

WTP_{ijt} = willingness of the ith individiual in the sample to pay for jth trajectory at time t

n_{jt} = number of individuals in the sample group bidding for jth trajectory at time t

N_{kt} = total annual visitation to the kth site or area at time t

The design of questionnaires to estimate willingness-to-pay by contingent valuation methods requires more precise characterization of consumer's surplus. Two measures of consumer surplus differ somewhat from the Marshallian measure discussed earlier (Hicks 1943). They are equivalent surplus and compensating surplus. Equivalent surplus is the payment, made or received, that brings the consumer to the subsequent level of welfare if a proposed change in the consumption of the good in question does not occur. Compensating surplus refers to the payment, made or received, to keep the consumer at his or her initial level of welfare if the proposed change does occur.

If the consumer has a "right" to the original situation, compensating surplus is the proper measure (Brookshire and others 1980). If the consumer has a right to the level of services after some change, the equivalent surplus would be the correct measure. The equivalent and compensating surpluses can be further referenced as willingness-to-pay and willingness-to-accept, depending on which reference point the consumer is entitled to.

For any given consumer, the willingness-to-pay for an increase in consumption (the compensated willingness-to-pay) will be exactly equal to his or her willingness-to-pay to avoid that reduction (the equivalent willingness-to-pay). Willingness-to-pay measures are, consequently, identical irrespective of whether they are compensating measures or equivalent measures (Randall and Stoll 1980). Willingness-to-accept measures are also identical irrespective of whether they are compensated or equivalent measures. The remaining issue, then, is whether willingness-to-accept measures differ significantly from willingness-to-pay measures.

The difference between willingness-to-pay and willingness-to-accept has been shown to be zero when there is no income effect (Randall and Stoll 1980, Willig 1976). Where income effects are present and the good is not easily divisable, the willingness-to-pay will be less than the willingness-to-

accept. The difference is a function of income, the price flexibility of income, and the Marshallian surplus.

The proper measure of valuation for wildfire impacts is the surplus measure of willingness-to-pay for two reasons. First, optimizing adjustments by the consumer are not possible with respect to increments or decrements in fire control and prevention services. Measures of variation are thus inappropriate. Second, while willingness-to-accept measures may be required to value decrements in resource flows, they are not likely to be easily obtained. The difficulty lies in the absence of real markets in which such compensating payments are made and the difficulties of establishing a contingent market in which the participants will regard compensation as a realistic and proper payment vehicle. In those cases, willingness-to-accept measures should be derived from willingness-to-pay measures (Randall and Stoll 1980). The sample interview sequence described earlier was designed to elicit ordinal preferences and measure compensated surplus.

The method discussed in this section should provide a workable framework within which the economc impact of fire on outdoor recreation can be assessed and evaluated. The method relies heavily on iterative bidding procedures which emerge as the only feasible means of assessing variations in the aesthetic attributes of environment. The procedure has been designed to permit testing for strategic and information bias. Hypothetical bias cannot be eliminated so possible effects stemming from the hypothetical nature of the procedure should be addressed explicitly in the analysis.

In applying the contingent market valuation method to assess preferences and willingness-to-pay of respondents to fire-affected sites, such respondents should be drawn from the relevant segments of the recreation population. Campers, sports fishermen, and hunters are examples of such segments. The example application of the approach to the 69 students displays that such future applications are feasible.

CONCLUSIONS

Inferential techniques for measuring consumer surplus rely on the observed behavior of recreationists to infer willingness-to-pay for their recreation experience. The most prominent and widely used of these techniques is the travel-cost demand methodology. Much of the data needed to utilize this approach is readily available, but travel-cost demand methods are not sensitive enough to measure the impact of fire upon recreation. They are highly aggregative and not well suited to measure the value of specific environmental attributes. Additionally, they cannot be readily used to measure time-dependent phenomena—and the effects of fire depend on time and on the type of environmental attributes.

Contingent market valuation methods—a direct, economic approach using personal questionnaires—appear suitable for fire-impact studies. These methods do not appear to lead to biases from strategic bidding behavior and selective provision of information if the characteristics of the contingent market are explicitly defined for the potential consumer. The hypothetical nature of such markets poses a continuing problem inasmuch as there is no way to test for such biases.

The measurement of the esthetic impact of distributed forest environments uses both economic and psychological methods. The most complete economic studies employ travel cost demand methods, but they fail to deal with substitution possibilities in a realistic way, and they neglect time-dependent impacts. Psychological studies provide some insight into the structure and determinants of preferences for disturbed landscapes, but they offer little promise for measuring preferences quantitatively.

Willingness-to-pay is the appropriate measure for valuing the impact of wildfire on perceptions of recreationists. Valid willingness-to-accept measures cannot be obtained because of the absence of real markets in which compensating payments are made and because of difficulties in establishing a contingent market in which the participants will regard compensation as a realistic and proper payment vehicle.

Equivalent measures of consumer surplus are appropriate for valuing the impact of wildfire on recreation. Such measures are based on the premise that the consumer cannot make optimizing adjustments in the consumption bundle in response to incremental changes in resource flows. This premise is consistent with the conditions under which fire does, or does not occur, in that fire prevention and control services are jointly supplied and consumers cannot adjust in response to changes in the level at which those services are provided.

A contingent market valuation approach was used to value the impact of fire on recreation. Respondents in the study were asked to express their preferences to a sequence of photographs. The results suggest that less intense fires may have beneficial economic effects on recreation, whereas intense fires may have adverse effects. The impact of fire is not always negative or unanimous, and preferences among recreationists may change over time. This difference underscores the need to account for vegetative succession trajectories in assessing the impact of fire. Although the contingent market valuation approach proved suitable in this study, an assessment of the impact of fire on the perceptions of different types of recreationists is largely a statistical problem in sampling. The problem should be addressed in a formal sampling design that would be associated with efforts to test the methodology empirically.

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The Forest Service, U.S. Department of Agriculture, is responsible for Federal leadership in forestry. It carries out this role through four main activities:

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The Pacific Southwest Forest and Range Experiment Station

- Represents the research branch of the Forest Service in California, Hawaii, and the western Pacific.

Vaux, Henry J., Jr.; Gardner, Philip D.; Mills, Thomas J. **Methods for assessing the impact of fire on forest recreation**. Gen. Tech. Rep. PSW-79. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1984. 13 p.

Methods for assessing the impact of fire on forest recreation were studied in a literature search and an experiment. Contingent market valuation appeared the most promising. This direct, economic approach uses personal interviews and sets up a hypothetical market transaction in which values are estimated. In an illustrative application of this method, respondents were shown sequences of photographs of recreation sites depicting scenes before and after a fire. They were asked about their preferences among the scenes depicted and about their willingness-to-pay an entrance fee to the preferred sites. The example results suggest that less intense fires may have beneficial economic impacts on recreation values, whereas intense fires may have adverse effects. The valuation of the impact of fire among recreationists is not always negative or unanimous and preferences may change over time.

Retrieval terms: recreation economics, travel cost demand models, contingent market valuation, wildfire impacts, landscape attributes, wildland fire management.

United States
Department of
Agriculture

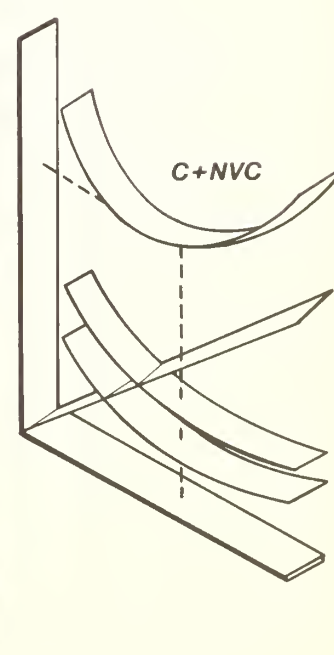
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-80

Risk in Fire Management Decisionmaking: techniques and criteria

Gail Blattenberger William F. Hyde Thomas J. Mills



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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

October 1984

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CONTENTS

About one-tenth of the Forest Service, U.S. Department of Agriculture's budget is allocated to fire management activities. In fiscal year 1979, the budget item for these activities totaled $216 million. Not included in this amount are the resource losses that result from wildfire. Such losses are always considerable in spite of eventual fire control (U.S. Dep. Agric., Forest Serv. 1980).

Fires are an even greater management problem than the budget figures suggest because of the difficulty of predicting their frequency, behavior, and effect on the natural resource. Factors that influence fire, such as weather and vegetation, are highly variable. Viewed in the context of decision theory, the fire management system is, therefore, pervaded by conditions of risk and uncertainty. As these terms are generally used, a *risk* exists when the potential outcomes of an action can be assigned probabilities; *uncertainty* exists when the probability of the varying outcomes cannot be determined (Knight 1933).

The data required to develop accurate probability information for the fire management system is difficult to obtain. In the past, fire managers have relied heavily on experience and intuition, leading to rules-of-thumb, such as the requirement adopted by the Forest Service in 1934 that all fires must be controlled by 10 a.m. of the second day if control is not possible during the first day. It seems likely now that decisionmaking based on analysis of risk and uncertainty inherent in a given fire management problem can increase the efficiency of fire control planning. Once risk is considered, decisions can be made in accordance with the risk preference of the planner and the public body which the planner represents. By risk *preference*, we mean willingness or unwillingness to take a risk.

Management strategies vary with attitudes toward risk, and these, in turn, vary with the scope of management responsibility. Management strategies also vary with the different components of a fire management program, from prevention and detection through fuel treatment and fire suppression activities. Finally, a manager of public lands must deal with the differences in risk and risk preferences as perceived by the manager, the fire management agency, and the public. Current rule-of-thumb strategies may be expedient in the short run and even cost-effective when decisions must be made in real time. For long-term fire program planning, however, a more constructive approach requires probabilistic analytical models, and full consideration of the risk consequences on alternative management actions.

A probabilistic fire management planning model would include all major sources of variation, from fire weather and fire occurrence through fire suppression effectiveness and resource output values. The risk embodied in a fire program option could be represented graphically and displayed in a number of ways. Whatever the approach, the risk should be explicit and should be weighted by the decisionmaker. It should not be embedded in the analytical model. For example, in the National Fire Danger Rating System, fire weather is measured only on exposed southwest slopes at 2 p.m. (Deeming and others 1977). A particular risk stance is thus determined by the measurement itself. Testing fire program performance against the 90th percentile burning index in the fourth worst year is another example (U.S. Dep. Agric., Forest Serv. 1977). Both of these measures imply a specific fire hazard which many fire control analysts believe is more severe than generally encountered.

This report examines economic theory as it is applied to fire management, describes a simulation model that includes the factor of risk, and compares fire management with other natural hazards to identify unique characteristics. Empirical data to implement the model immediately do not exist because measurement techniques and data collections are not yet adequate for that purpose. The model can suggest directions, however, for future study and identify questions that should be answered.

ECONOMIC THEORY OF FIRE MANAGEMENT

The most commonly discussed framework for economic analysis of fire management is a model that demonstrates the outcome of fire programs as their cost plus the fire-induced net value change in natural resource outputs. The fire program which minimizes "cost plus net value change" (C + NVC) is also the program which maximizes present net value (Mills 1979). Net value is appropriate here because fire effects can be beneficial as well as detrimental, as in their typical influence on wildlife habitat. The C + NVC model is a modification of one developed more than 50 years ago (Headley 1916, Sparhawk 1925). Fire suppression costs and net value changes due to fire, both beneficial and detrimental, diminish as presuppression expenditure increase (program level PL) (*fig. 1*). The C + NVC curve is the simple sum of two cost and two net value change curves. PL_1 is the level of presuppression expenditure at the minimum C + NVC, and therefore is the most economically efficient expenditure level.

Determining the most efficient presuppression expenditure has not always been the primary purpose of the model. Variations of it use such measures as acres (Hornby 1936), suppression force size (Arnold 1950), and fire management effort index (Simard 1976). In addition, fire control actions can be varied in the analysis. Although these actions may be broadly classed as presuppression, detection, and suppression, distinctions between the variables or fire management program components are vague and general. Regardless of how the horizontal axis is labeled, however, the model is depicted as in *figure 1*. The impact of

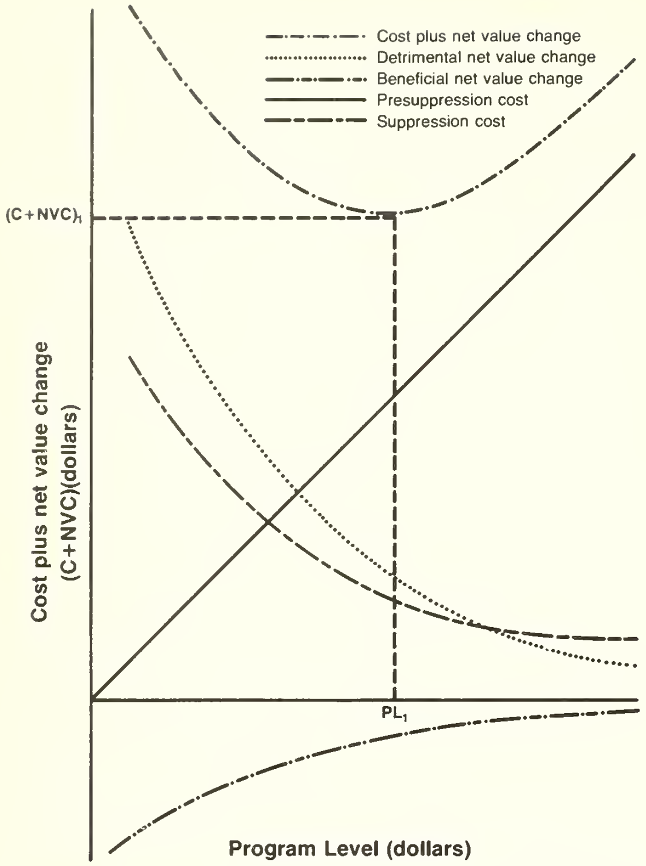

Figure 1—Suppression costs and changes—whether beneficial or detrimental—due to fire diminish as presuppression expenditures increase, according to this model of cost plus net value change.

increasing the fire control activity is a reduction in detrimental fire effects and suppression expenditure.

A review of the fire economics literature (Gorte and Gorte 1979) found little empirical application of the C+NVC model. More recently, however, analyses by Schweitzer and others (1982) and the U.S. Department of Agriculture, Forest Service (1980) applied the model to alternative National Forest System presuppression programs. A Forest Service analysis of presuppression programs on state and private lands has also been completed (U.S. Dep. Agric., Forest Serv. 1982b).

However the C+NVC model has been presented in the literature, the issue of risk is either short-changed or ignored. Variation in outcome, and therefore in the total C+NVC curve, is generally not considered. The fire economics literature which evaluates the proposed models (Davis 1965, Zivnuska 1972) shows awareness of risk, and Schweitzer and others (1982) touched on the issue by evaluating 3 years of varying fire season severity, but the risk-oriented analytical work is sparse. There is general agreement that any comparison between fire controls must consider its effect on outcome probability as well as its expected value economic efficiency. To make such comparisons possible, risk must be incorporated into the model. Before demonstrating how this can be done, however, we need to consider some theoretical principles of risk and uncertainty.

RISK AND UNCERTAINTY IN DECISION THEORY

In its simplest form, the decisionmaking problem is a choice among a set of actions. If all the conditions associated with the action are known and not subject to change, or totally within the decisionmaker's control, then the outcome of any action is a certainty, and the choice is made solely on the basis of the outcome desired. (We assume here that the conditions themselves are not affected by the choice of action). Normally, however, the outcome depends not only on the action chosen but also on unknown factors that are not in the individual's control. Each combination of these unknowns may be called a "state of nature." Based on two possible actions (a_1, and a_2) and two possible states of nature (s_1, s_2) there are four possible outcomes or consequences (C_{11} through C_{22}):

States:	Actions a_1	Actions a_2
s_1	c_{11}	c_{12}
s_2	c_{21}	c_{22}

In a state preference model, the planner's beliefs as to the likelihood of each state of nature are summarized as P_1 and P_2 *(table 1)*. These probabilities do not depend on the choice of action. The consequences multiplied by the state probabilities give results which, when summed for each action, provide a measure of the expected values v_1 and v_2 of taking the actions. The planner can compare these expected values, representing the long-term average outcome. The choice is straightforward if the planner simply selects the action with the greatest expected value, that is,

Table 1—*State preference representation of a decision problem*

Alternate actions	States of nature	Possible consequences	X	Probability of state	=	Expected value of action[1]	Variance
a_1	s_1	c_{11}		P_1		v_1	σ_1^2
	s_2	c_{12}		P_2			
a_2	s_1	c_{21}		P_1		v_2	σ_2^2
	s_2	c_{22}		P_2			

[1]Products of c_{ij} and P_j for each state are summed to equal v_i for each action.

with maximum utility. By the term "utility" we mean the value that the decisionmaker places on the outcome.

Although in this simple model the actions, states, and consequences are shown as discrete, each may be a continuum of values—that is, each may in turn be probabilistic. The states of nature in the fire management system are described by conditions such as weather events and fire occurrence frequency. The actions specified may include fuel treatment actions, suppression strategies, or a mixture of several different program components. The C+NVC consequences, being contingent on probabilistic elements, are also probabilistic.

In applications of the state preference model under risk conditions, the outcome probabilities for each state of nature are known with some degree of precision. Under uncertainty, however, the decisionmaker cannot determine the outcome probabilities. Various decision rules or strategies have been proposed for use under uncertainty, such as the minimax, maximax, or principle of insufficient reason strategies. Because these tend to be inconsistent or unsatisfactory, it is probably more useful to approximate risk conditions whenever possible than to apply uncertainty decision rules. That is, it is helpful to assign some probabilities to the various states of nature, even if these must be arrived at subjectively. For example, moderate to broad resolution estimates of fire behavior and fuelbed conditions were found to be adequate for many data needs in fire planning in a study made on a National Forest (Barrager and others 1982). Highly refined data are not always necessary. For this reason, this discussion of decision theory will consider risk conditions rather than uncertainty.

An action with a certain outcome is a useful benchmark for evaluating the individual's view of the risk associated with other actions with probabilistic outcomes. Thus, if the probabilities for s_1 are 100 percent, those for s_2 must be 0, and the choice presents no risk. If the consequences c_{11} and c_{21} are identical (action 1 produces the same result under either state of nature), the outcome of action 1 is certain and without risk. Decisionmakers may show indifference between these actions whose outcomes are certain and others with probabilistic outcomes. If they do, the outcome of the certain action is known as the *certainty equivalent*, and it can be used to measure their degree of risk aversion, that is, their unwillingness to take an even bet.

One way to represent degrees of risk aversion that an individual displays using the certainty equivalent is to depict utility functions *(fig. 2)*. The horizontal axis, which measures the variance of outcome, represents the degree of risk. Here v_1 is the mean value of a particular event whose outcome is certain (has a zero variance). If the decisionmaker shows indifference between it and the expected value v_2 of an event whose outcome carries risk (σ_2^2), then the individual's *risk premium* is $v_2 - v_1$. That is, the individual will accept greater variance in the expected outcome only if the outcome has a greater expected value. Each curve traces a locus of such points, called the decisionmaker's *indifference curve*. The parameter (here variance) over which risk preference is shown may take other forms. This general approach for comparing two parameters is termed "parameter preference" (Tobin 1958).

In many specific situations, the decision reached on the basis of parameter preference may be the same as that which simply

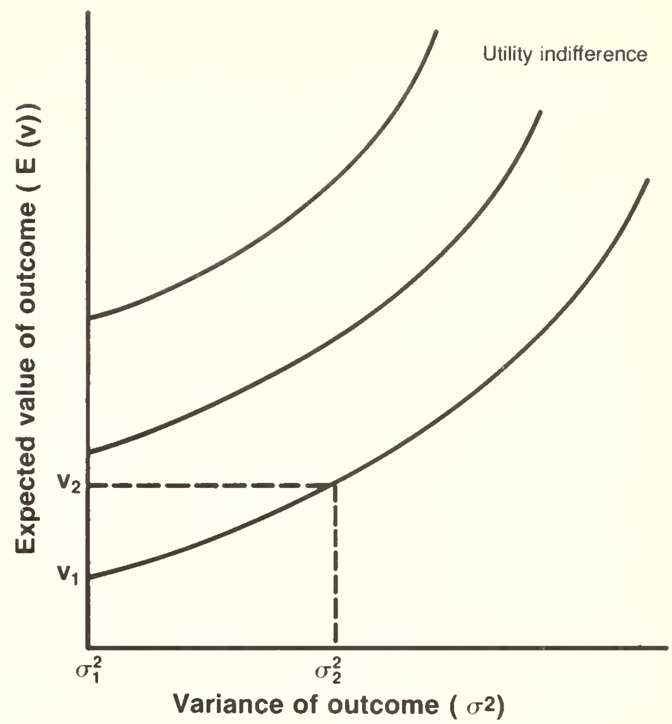

Figure 2—The parameter preference model for mean-variance makes the trade offs between expected value return and return variability explicit.

maximizes the expected value outcome as shown in the state-preference model. In others this is not true *(table 2)*. Each action in this example has the same expected value, and the same variance. Risk-averse individuals confronted with this choice, however, would normally select action a_2 to avoid the possibility of the -8000 consequence, a decision not accounted for by simple parameter preference.

DECISIONMAKER'S APPROACH TO RISK

The attitude of the decisionmaker with respect to risk may range from risk aversion through risk neutrality to risk preferring. In general terms, the purchase of an insurance policy is an illustration of risk aversion; the difference between purchasers'

Table 2—*Example of ambiguity in application of parameter preference to a decision problem*

Alternative actions	Consequences X	Probability of states	=	Expected value	Variance of expected value
a_1	c_{11} = 2,000	0.9		1000_1	9×10^6
	c_{12} = -8,000	.1			
a_2	c_{21} = 0	.9		1000_2	9×10^6
	c_{22} = 10,000	.1			

[1]Products of actions a_1, a_2, and states s_1, s_2 *(table 1)*.

expected loss and their insurance cost is their risk premium. Gambling, in most instances, illustrates risk preferring. The risk neutral decisionmaker is indifferent to risk and considers only the expected value outcomes in the decision.

Risk Aversion

Risk aversion—the unwillingness to accept a fair bet—can be illustrated *(table 1)*. If the consequences of action a_1 are a win (W) under state s_1 and a loss $(-L)$ under state s_2, and those of a_2, an insurance action, are a risk-free cost (D) in both states, then we can express v_1, the expected monetary value of each, in this way:

$$v_1 = (P_1W) + [P_2(-L)] \text{ and}$$
$$v_2 = P_1[W + (D\text{-}W)] + P_2[-L + (D + L) = D$$

A risk-averse decisionmaker would choose action 2, even though its monetary value was less than that of action 1. The risk premium paid for the risk-free insurance option would be $v_1 - D$.

In the real world, fire managers and public fire management agencies appear to be risk averse. It is reasonable to assume that local fire program decisionmakers are more risk averse than their agencies or than society as a whole because they see more localized sets of cost figures and benefits. To decisionmakers, the probability of adverse outcome is low but its relative impact is great.

Particular conditions may increase risk aversion. Consider, for example, foresters with management experience in various nonfire assignments who are transferred to fire management positions. They may expect their new jobs to be their only fire experience, and they also may expect that their future careers depend on there being no large fires during their tenure. The utility curve will, therefore, reflect acres burned instead of economic value. And that utility curve will also show a greater degree of risk aversion than that for foresters with more fire management experience who intend to stay in a fire position, because foresters with less experience are probably less sure of themselves, and their personal costs of failure are greater.

The prevalence of risk aversion has led to the development of economic institutions for shifting risks, such as market insurance and the issuance of common stocks. Risk sharing is also a response to risk aversion; movement of crews and equipment across public agency boundaries is an example of risk sharing in fire management.

Even in the presence of risk aversion, however, universal full insurance is virtually impossible. Budgets limit the amount of insurance that can be purchased. There is also a condition known as *moral hazard* that hampers full risk sharing. Issuance of insurance can affect the incentives, which in turn affect the probabilities and outcomes, and therefore the risk itself. This is why private timber companies cannot obtain full insurance against forest fires; it would decrease their incentive to control fires.

Risk aversion enters strongly into investment decisionmaking since investment is the exchange of certain returns now for probabilistic returns in the future (Hirshleifer 1965). This is particularly true of long-term investments, such as those typical of natural resource production processes. In the absence of perfect markets, risk averse individuals may discount probabilistic returns at a higher rate to accommodate risk.

The implications of risk aversion for public investment have been widely discussed. Some economists have contended that public agencies should be risk-neutral because of the diversity of their investments (Samuelson 1964), and because risks are diluted by spreading them along a large number of actors (Arrow and Lind 1970). That is, public agency budgets are large and relatively unconstraining; therefore, agencies with many investments can self-insure against most risks. This approach explains arguments for a risk-neutral fire plan for the National Forest System. Budget levels probably do not limit self-insurance of large timber companies from fire losses, either. Thus, risk neutrality may be justified for both public agencies and large private firms.

Arguments for risk-neutral decisions disregard several factors, however. Moral hazard is one; another is the distinction between institutional objectives and the personal objectives of individual managers. Fire managers whose careers depend on the outcome of their decisions are likely to be risk averse. A third factor is the uneven distribution of benefits and costs. Risk neutral decisions assume that these are uniformly distributed and independent of the states of nature, an assumption that is unrealistic for most public investments, particularly those involving natural resources. Fire protection is more beneficial in dry southern pine forests than in the spruce-fir bogs of Maine. National forest fire protection is also more beneficial to local communities dependent on timber for their livelihood than it is for the general taxpayers who are lumber consumers.

Decision theory is, therefore, relevant to forest fire management. Wildfire management consists of adaption to risk as well as risk sharing. Productive adaptations, like fire roads and fuels management, can change the total risks to be distributed, but risk aversion is still important in local fire management.

The following simple example clarifies some of these concepts. A fire manager confronted with a wildfire may have two alternatives: a_1, monitor fire progress while allowing it to burn, or a_2, take aggressive measures to suppress it. A low intensity fire can have beneficial effects for wildlife habitat and grazing. The outcome under each action and state of nature are these:

	Values for actions		
States:	a_1	a_2	
(Fire intensity)	(monitoring)	(aggressive suppression)	Probabilities
s_1 (high)	$10,000	$1,200	0.1
s_2 (low)	−1,000	−900	.9

A positive C+NVC is a net loss, therefore, the expected C+NVC outcome is 100 for a_1 and −690 for a_2. The expected monetary C+NVC's of the two actions might lead the manager to choose the monitoring action. The risk-averse manager, however, may prefer the more certain returns of the aggressive suppression action. The manager's implicit risk premium is at least $790.

Table 3—*Example of a paradoxical group decision based on probabilities and utilities as seen by two risk-neutral individuals*

Decisionmaker	Action	State of nature	Utility[1] (outcome)	Probability	Expected value
Timber company manager	No prescribed burn	Wildfire	9500	0.1	950
		No wildfire	0	.9	
	Prescribed burn	Wildfire	1000	.1	
		No wildfire	1000	.9	[2]1000
Wilderness guide	No prescribed burn	Wildfire	7000	.3	980
		No wildfire	-1600	.7	
	Prescribed burn	Wildfire	1000	.3	
		No wildfire	1000	.7	[2]1000
Group	No prescribed burn	Wildfire	8250	.2	
		No wildfire	-800	.8	[2]1010
	Prescribed burn	Wildfire	1000	.2	
		No wildfire	1000	.8	1000

[1]Expected monetary value of the cost plus net value change. Positive benefits are shown as negative values on this basis.
[2]Action with the greater expected value.

Group Decisions

Degree of risk preference enters into group decisions in complex ways. Members of the group may differ radically in their preferences and their assessment of the states of nature. For example, fire roads may be desirable to those who want fire control that is aggressive enough to exclude fire from an area, but may be obstacles to those who prefer wilderness preservation.

The paradoxes that arise in group decisionmaking have been described by Raiffa (1970). Consider a choice of whether to do a prescribed burn. A probability exists that a wildfire could occur after the prescribed burn. The probabilities of subsequent wildfire and the utilities as seen by two advisers—one a corporate timberlands manager and the other a wilderness tour guide—can be contrasted, along with the resulting group decision *(table 3)*. For simplicity, both advisers are shown as risk-neutral. Considering a wildfire unlikely, the timberlands manager prefers no prescribed burn. The guide also prefers no burn because the consequences of a wildfire when there has not been a previous prescribed burn are relatively less severe, and there are positive benefits to no fire at all planned or unplanned. A group decision here was obtained by weighting the utilities and probabilities of the two advisers equally. The group decision, based on expected value outcome, favors the prescribed burn. The group decision is the reverse of the individual decisions. This may be a rational choice if the probability assessments are independent of the assessments of the consequences.

INCLUDING RISK IN C+NVC MODEL

The risk aversion that we suggest is present in fire managers and public fire agencies strongly implies that risk must be evaluated and incorporated into policy models such that the trade off between risk and efficiency can be clearly defined. This can be accomplished by including probabilistic considerations and outcomes within the C+NVC model output.

In depicting the model, acres burned is the surrogate measure of fire effects *(fig. 3A)*. The curve shows the expected effects of fire management effort on acres burned as they are frequently reported in the literature. The curves at PL_1 and PL_2 are the probability distributions of acres burned at those funding levels. The increased expenditure, then, reduces the expected acres burned and reduces the width of the probability distribution. The risk-averse fire manager will expand the fire management effort beyond the point where its cost equals the reduction in loss.

In *figure 3B*, the probability distributions are shown in relation to the C+NVC curve in *figure 1*. Recalling our discussion of state preference, we assume that the fire manager's problem is to maximize net utility, which is assumed here to be the same as minimizing C+NVC, if risk neutrality prevails. In *figure 3B*, as expenditures on presuppression inputs increase, the probability functions become tighter and risks decrease. Risk averse man-

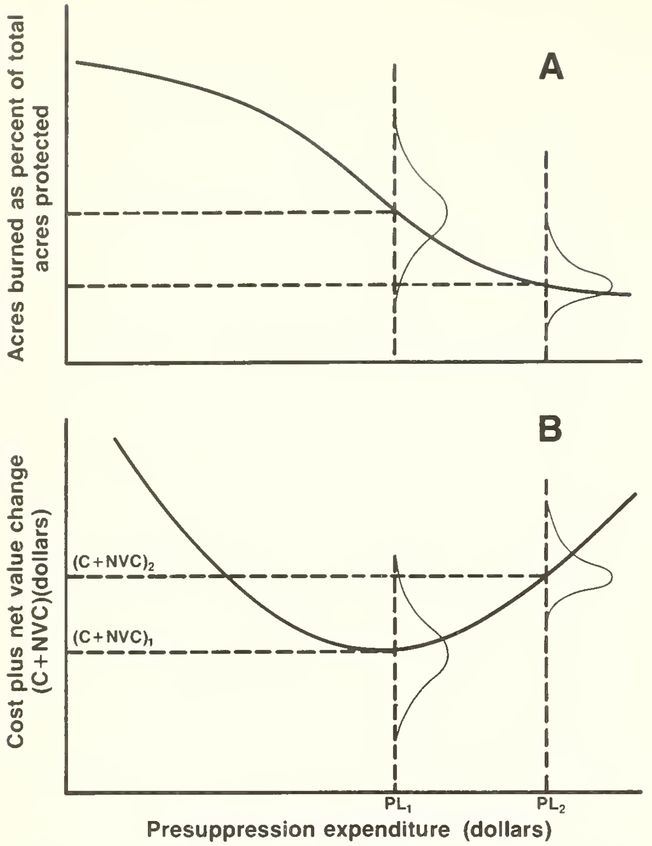

Figure 3—The expected reduction in cost plus net value change variation with increasing presuppression expenditures must be compared with reductions in expected value efficiency beyond the most efficient program level.

agers will extend presuppression effort beyond PL_1, the least cost plus solution. If PL_2 is the chosen level of presuppression expenditure, then $(C+NVC)_2-(C+NVC)_1$ is the risk premium associated with the reduced risk. Although this model is purely theoretical, it provides a framework for the consideration of data and measurement problems.

DATA AND MEASUREMENT

For eventual development and implementation of a decision-making model that includes a consideration of risk, empirical data must be gathered and analyzed. Certain critical areas of the fire control management problem will need to be considered, such as:

- What are the temporal and spatial units that define the problem?
- What is the appropriate utility curve for fire management?
- What are the control and state variables?
- What are the effects of management actions?

In addition, the probabilities of state variables and the effects dependent on them will need to be estimated.

The measurement units generally selected for time and space are the annual fire season and the fire management area. Annual evaluation of fire damage and fire management is reasonable. It seems important, nevertheless, to include multiple season fire management effects in the annual analysis because there are correlations between fire risk and damages from year to year. The effect of scale may be relevant; the variance in fire effects is reduced over longer time periods, and there are short periods of high risk when fire danger is great.

The fire management area is a more ambiguous unit. It includes land under the jurisdiction of a fire manager for fuels management, prevention, detection, and suppression. It is often determined by physical barriers, but in the absence of these, the fire manager must plan for possible fire spread to adjacent management areas. Again, the scale on which decisions are being made affects risk assessment in fire management. A local fire manager is likely to be more risk averse than one setting national fire policy. The Chief of the Forest Service can accept a single 30,000-acre fire much more easily than a District Ranger. The probability of such a fire somewhere within the National Forest System is much greater than that on a given District, but its effect is much smaller. Therefore, some risk-related policies, such as prescribed burning, might be less attractive to a local fire manager than to a planner at the national level. This fact must be considered if broad social objectives are to be satisfied in public land management planning.

The major utility problem to be solved in natural resource areas is the difficulty of assigning value to the multiplicity of resource outputs that originate on forest land. Many, such as water and range, are not exchanged freely in the market, and some, such as certain types of recreation, are not exchanged at all. Fire managers must also show concern for such values as public safety, air quality, and soil stability.

Additional problems are presented by the fact that the impact of wildfire varies with the character of the fire and the fire site, with the availability of substitute resources, and over time after the fire. Also, fire damage is not always restricted to the fire area, but may include offsite effects, as flooding. The intensity of the fire as well as acreage burned is important, and is not always constant over the total fire area. Low intensity fires can have positive effects, even to the extent of producing a positive net value change in resource output, either initially or several seasons after the fire.

What is the appropriate utility curve for a public land management agency? Perhaps the damages of small fires can be ignored. If large fires are truly unlikely, then they too can be ignored in the general daily management plan and in the evaluation of a local manager's performance. Perhaps the severest events cannot really be planned for in a presuppression sense. Fire managers should concentrate their fire management skills on the broad midrange of all potential fires. Specifications of the utility curve and the state variables with their probabilities could set bounds on the fire sizes that should be subject to management strategy.

Measurement of fire control and state variables is an easier task than determining utility. Control actions are commonly separated into fuels management, prevention, detection, and suppression variables. Inputs of personnel and equipment within

these classes are readily measurable in both physical terms and dollar value. There are also common measures of important state variables, such as wind, weather, fuel volume, and fine fuel moisture content.

Evaluating the outcome of fire management actions presents more serious problems because we must estimate the probability of their various outcomes. The probability of both lightning- and human-caused fire occurrence has been investigated and its distribution has been closely approximated by a Poisson probability function (Cunningham and Martell 1973, Fuguay 1978). Fire spread (Rothermel 1972) and probability of ignition have also been studied in relation to state variables, but unfortunately there has been little empirical work tying these probabilities to the fire control variables. Initial attack effectiveness has been modeled (Albini 1976, Bratten and others 1981, U.S. Dep. Agric., Forest Serv. 1982a). Fireline construction rate estimates vary considerably, however, and the variability can influence effectiveness of both initial attack and large fire suppression actions (Haven and others 1982). These relationships must be better understood if fire controls are to be evaluated with respect to risk.

These comments on data and measurement problems are presented in terms of the model presented earlier. Considerable amounts of additional data are needed before the model can be fully developed empirically. Even with present data, however, empirical application can be made to small subsets of the forest fire problem.

RISK AND HAZARD MANAGEMENT

Fire is only one of the natural hazards that managers of both public and private lands must address. A review of the characteristics, costs, damage, and control timing of snowstorms, floods, oilspills, and forest fires permits us to identify unique features of fire management *(table 4)*. It would be instructive to make a similar analysis for a more complete list of natural hazards, including earthquakes, volcanos, hurricanes, urban fires, droughts, and others.

The damage characteristics of natural hazards are based on a combination of the hazardous event and the existing state of human adjustment. The local inhabitants of the area north of the Arctic Circle have adjusted thoroughly to snowstorms, events which are hazardous but seldom catastrophic. Where hazards tend to be catastrophic, development is limited, so that damage is held at a low level. The potential for nuclear accidents to be catastrophic explains why nuclear facility designers display greater risk aversion than do planners for controlling the various damaging but noncatastrophic natural hazards. (Starr 1969).

Forest fires resemble other hazards, but also differ from them. Several features of forest fire damage distinguish it from other hazardous events. First, the potential fire damage is complex, because of the multitude of goods and services originating in forests. Second, fires can alter the array of these goods and services for a number of years, with the unique characteristic that the alteration may be beneficial.

Table 4—*Characteristics, costs, damage, and control timing of some unavoidable natural hazards*

Feature	Hazard
	Snowstorms
Characteristics	Recurring, seldom catastrophic, not preventable, but can be forecast
Costs	Long-run investment in equipment and manpower, short-run cost for immediate response
Damage	Variable, short-lived
Control timing	Before event: minimal (weather monitoring, structural modification) During and after event: snow removal
	Floods
Characteristics	Recurring, seldom catastrophic, not preventable but can be forecast; also control of development of floodplain
Costs	Long-run investment in equipment and manpower, short-run cost for immediate response; also dams, levees, and costs of development foregone
Damage	Variable, relatively short-lived
Control timing	Before event: weather forecasts, reports of upstream river conditions During and after event: some levee building, rescue and relief, cleanup
	Coastal oilspills
Characteristics	Recurring, rarely catastrophic, preventable (avoid development)
Costs	Cost of opportunity foregone; short-run cost for cleanup equipment and manpower
Damage	Variable, some immediate, some potentially lasting
Control timing	After event: cleanup
	Forest fires
Characteristics	Recurring, seldom catastrophic, some preventable
Costs	Long-term investment in equipment and manpower, short-term cost of immediate response
Damage	Variable, can be long-lasting, may be ultimately beneficial
Control timing	Before, during, and after event: fuel modification, suppression activities, cleanup and salvage

In general, controls for most hazards concentrate on one or another of the three control periods specified in *table 4* (before, during, or after the event has occurred). Secondary controls are often applied at a different time than primary controls, however. The third unusual feature of fire management is that available actions are unusually varied and may occur in any or all three periods, before, during, and after the hazard event. Prevention, fuels management, weather forecasting, and presuppression activities occur before fire ignition; detection and suppression occur during the fire; and timber salvage and erosion control occur after the fire. Activities affect the likelihood and intensity of fire, and the magnitude of the damage. Suppression currently receives the bulk of attention, but fuels management is growing in importance.

While this diversity of control options does not suggest that forest fire managers can ignore risk, it does show there is greater flexibility for managing the risk than is typical of other hazardous events. The long-term forecasts and heavy capital investment nec-

essary in flood management, for example, which suggest early and strongly risk averse decisions, are largely absent. Nor are fire controls as simply reactive and delayed as are snow removal actions. Greater variety in control options enables fire managers to operate closer to their mean expectations of damages due to the event;.that is, to be less risk averse. It also requires careful evaluation of trade offs between the various control options.

Any comparison of forest fire control with control of other natural hazards must consider how decisionmakers plan for the riskiness of various perils. Where probability distributions and economic costs and damages are uncertain, we find general reliance on ad hoc criteria that imply an acceptable level of risk. Pollution and flood control planning can be examined to determine how these criteria compare with those common in fire control.

Congress has applied three different canons in legislation on the effects of environmental pollution on human life. The 1963 Clean Air Act applies a *health-based* standard, by which the maximum acceptable pollutant level is that at which the population group most sensitive to air pollution is not affected; this is reduced by a "margin of safety" to protect against undetected dangers (Thompson 1979). Such a standard implies either an infinite value on human life or the most extreme form of risk aversion. The 1972 Federal Water Pollution Control Act applies a *technology-based* standard, by which technological and economic feasibility dictate level of abatement (LaPierre 1977). This imples a greater acceptance of risk, as industries have little incentive to produce new technologies to clean up their wastes. The third canon is a *combination* of the two. The 1977 Noise Control Act and the 1974 Safe Drinking Water Act require consideration of both public health and economic-technical feasibility (LaPierre 1977). Most legislation takes this intermediate course, but the emphasis either way varies.

Legislation affecting flood planning takes a different tack. Federal funding encourages preparation of local disaster plans for such events as hurricanes, which lead to flooding at the time of the storm. In addition, the 1968 National Flood Insurance Act makes insurance available to communities with development guidelines for flood-prone areas. Recent amendments also prohibit Federal funds from supporting uninsured development in such areas. The decrease in development incentive decreases risk of losses from flooding (Baker 1979).

Similar standards have been employed with fire. An early Forest Service example is the 1934 policy to control all fires during the first burning day which avoids the risk of less favorable weather conditions the next day. Another early standard was the Forest Service's 1972 planning criterion of building a presuppression program capable of suppressing fires that occur under specified conditions (the 90th percentile burning index that occurred in the fourth worst of the last 10 years) at 10 acres or less (U.S. Dep. Agric., Forest Serv. 1977). The Forest Service fire management policy was revised in 1978 to mandate that the fire management program should be "cost efficient and responsive to land and resource management goals and objectives" (U.S. Dep. Agric., Forest Serv. 1983). The corresponding fire suppression policy states that the selection of suppression actions must consider cost effectiveness and probabilities of success.

Although the revised fire management policies and standards are not as concise and unambiguous as the earlier physical fire standards, they require a more thorough evaluation of the trade offs between risk and other objectives, such as the cost-effective contribution to land and resource management objectives. The presence of data and measurement problems and an incomplete model design are obstacles to such risk evaluation, but previous studies (Schweitzer and others 1982) demonstrate that some quantitative risk analysis is possible. The design of a probabilistic fire management analysis model by Bratten (1982) and Mills and Bratten (1982) provides further evidence that fire risk evaluations are feasible.

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In the past, decisionmaking in wildland fire management generally has not included a full consideration of the risk and uncertainty that is inherent in evaluating alternatives. Fire management policies in some Federal land management agencies now require risk evaluation. The model for estimating the economic efficiency of fire program alternatives is the minimization of the sum of fire management cost plus the fire-induced net value change in the resource outputs (C + NVC). Risk can be graphically displayed in this model by the addition of probability distributions about the C + NVC curve. Risk aversion, rather than risk neutrality or risk preference, is widely practiced in fire management, but attitudes toward risk can vary. Managers have greater flexibility in managing fire hazards than other natural hazards because their actions can take place before, during, and after the hazard event. This flexibility enhances their ability to be risk-neutral toward the fire hazard. Although data and measurement problems exist and model design is incomplete, an analytical evaluation of risk inherent in fire management alternatives is feasible.

Retrieval Terms: economic efficiency, expected value decisions, probabilities, risk, state preference, uncertainty

United States
Department of
Agriculture

Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-81

Regeneration of Douglas-fir in the Klamath Mountains Region, California and Oregon

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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

December 1984

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CONTENTS

One of the most valuable timber species in the United States is the magnificent coast Douglas-fir *(Pseudotsuga menziesii* [Mirb.] Franco var. *menziessi)* of western Washington, Oregon, and northern California *(fig. 1)*. The Pacific Douglas-fir forest cover type (Eyre 1980), with its associated species, is one of the world's most productive (Williamson and Twombly 1983). The preeminence accorded coast Douglas-fir is attributed to its fast growth, long life, and large size at maturity, and to the excellent structural properties of its wood. Considerable attention has been devoted to studying the regeneration of coast Douglas-fir to assure production of future crops.

Historical documentation of the evolution of logging practices and silvicultural methods in the Klamath Mountains Region of Oregon and California is incomplete, although such information would help provide a better understanding of how current regeneration practices developed. However, for three National Forests (the Six Rivers, Klamath, and Trinity) in northwestern California, some of this information is available. Cutting began there in 1947, but logging activities were minor before 1950. Heavy cutting did not start in most working circles until after 1955 (Hopkins 1964). The silvicultural system for Douglas-fir evolved from individual tree selection, to group selection, to clearcutting in blocks.

In the early 1950's, group selection was begun on the Klamath National Forest, and shortly thereafter on the Six Rivers and Trinity National Forests. The objectives of this cutting method were to harvest the most defective overmature trees, create openings large enough to minimize logging damage and competition to the future stand, and yet keep them small enough to promote natural regeneration.

During these first years of harvesting in northwestern California, all yarding was by tractors. On steep slopes this technique caused considerable soil movement, so the advantages and practicability of cable logging were explored. Because (for economic reasons) cable logging requires larger volumes to each landing, clearcutting in blocks was introduced. Between 1955 and 1960, the Six Rivers, Klamath, and Trinity National Forests, in that order, began administering sales that required cable methods on some areas.

After inspecting older cuttings in Douglas-fir and being briefed on the latest research findings, the Forest Supervisors of these three forests, in 1958, agreed that natural seeding was not providing adequate restocking. They decided to plant bare-root seedlings on all clearcut blocks as soon as possible after slash disposal. The amount of Douglas-fir planted on cutover areas had been small before then because natural regeneration had been expected.

First-year survival of Douglas-fir stock was poor in the early plantings, ranging from 0 to 70 percent, but generally less than 40 percent. Since then, planting stock, planting practices, and field survival have improved considerably. However, plantings can still fail because of adverse weather or harsh sites, but more often because of human error.

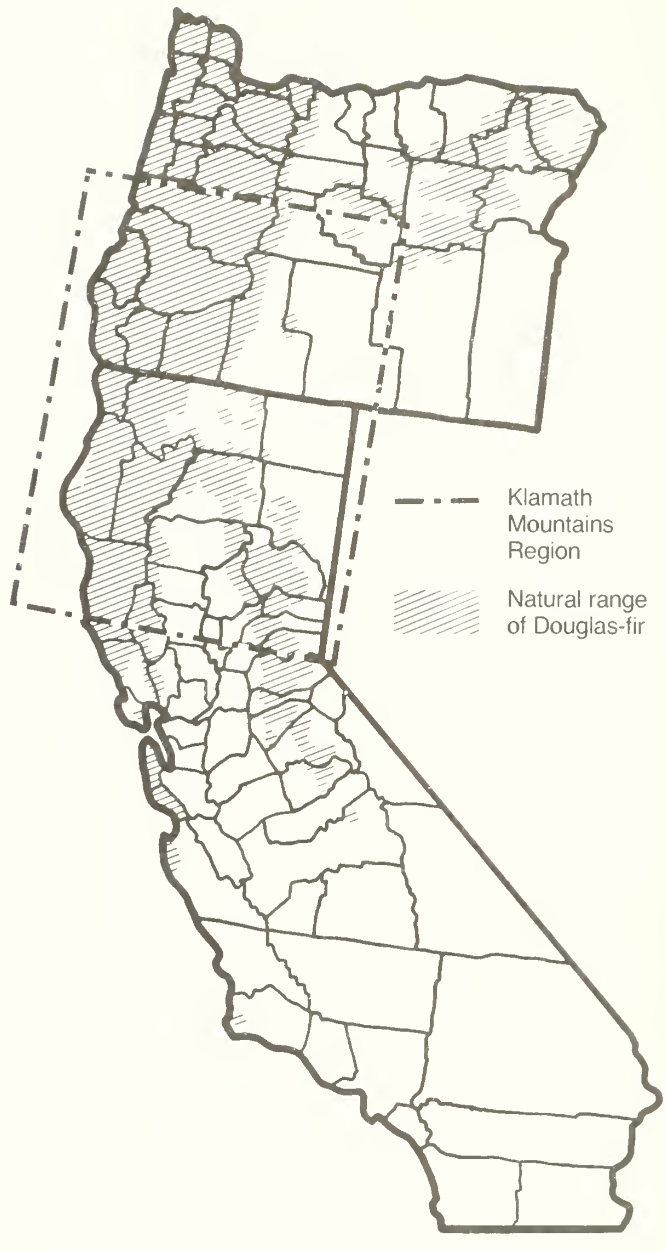

Figure 1—The Klamath Mountains region extends from southwestern Oregon to northern California (Little 1971).

Regeneration of Douglas-fir is not a problem on all sites, or throughout its range. On the cooler, moister coastal sites, it can be regenerated more easily than on the hotter and drier inland sites. In a similar manner, the species is regenerated more easily near the northern end of its range than

near the southern end. For these reasons, the general physical characteristics of the Klamath Mountains Region must be understood. However, numerous factors other than moisture and temperature are involved in regeneration success.

The regeneration process in Douglas-fir normally follows this sequence: seed production, site preparation, and use of one of the three principal regeneration methods—natural seeding, direct or manual seeding, or planting seedlings. Young stands must be carefully tended because the whole regeneration effort can be thwarted if competing vegetation or foraging animals are allowed to suppress or destroy the young seedlings.

This paper summarizes information about the regeneration of Douglas-fir in the Klamath Mountains Region of northwestern California and southwestern Oregon. Research in the adjacent Coast Ranges of the two States is included as well as relevant studies done elsewhere. This report is designed to serve as a convenient reference, by drawing together the findings of many studies, including some not previously published. And it offers general recommendations for the forest manager and other practitioners.

KLAMATH MOUNTAINS REGION

Geology, Topography, Soils

Excellent descriptions of the geology, topography, and soils of the Klamath Mountains and Coast Ranges are offered by Irwin (1966), Irwin and Hotz (1979), Page (1966), and Snavely and MacLeod (1979).

The Klamath Mountains province of northwestern California and southwestern Oregon, an elongate north-south area of approximately 312,000 ha (12,000 square miles), is bordered on the east by the Cascade province, on the southeast by the Great Valley province, and on the west by the Coast Range province *(fig. 2)*. The terrain in this province is highly dissected and mountainous, with ridge crests 1,500 to 2,100 m (5,000-7,000 ft) above sea level and peaks as high as 2,700 m (9,000 ft). Slopes are steep, and relief is commonly 900 to 1,200 m (3,000 to 4,000 ft) in many stream canyons. Evidence of glaciation is widespread in the high mountains. Vestiges of an old erosion surface are recognizable at many places, especially in the western part of the province, where many ridges are approximately the same altitude and the dissected remains of broad valleys are visible, suggesting that relief was generally low before the region was uplifted and dissected by the present drainage system.

The drainage pattern is complex, but the main streams generally flow westward, transverse to the lithic and structural grain of the province. The southeastern part of the province is drained mainly by tributaries of the south-flowing Sacramento River. The remainder of the province is drained mainly by the very extensive Klamath-Trinity River system in California and the Rogue River system in Oregon. Streams in the extreme northern part of the province are tributary to the South Fork of the Umpqua River.

In contrast, altitudes in the Coast Range province are lower, and only a few peaks, along the divide between the northern Coast Ranges and Sacramento Valley, are as high as 1,800 m (6,000 ft) and show evidence of former glaciation. Accordant ridges are common, as in the Klamath Mountains, but generally lower. Drainage patterns of the principal rivers of the northern Coast Range, such as the Eel, Mad, and Van Duzen, tend to parallel the northwesterly structural and lithic grain of the province. Exceptions are Klamath, Rogue, and other streams that rise in the Klamath Mountains and cut across the Coast Ranges on their way to the ocean.

Rocks of the Klamath province consist mainly of marine volcanic and sedimentary rocks that range in age from early Paleozoic to middle Mesozoic and are part of a sinuous belt of old rocks exposed also in the Sierra Nevada of California and in the Blue-Ochoco Mountains of east-central Oregon. For the most part these rocks have been extensively folded and faulted, weakly to strongly metamorphosed, and intruded by many bodies of granitic and other plutonic rocks. Ultramafic rocks, mostly serpentinized peridotite, are plentiful—indeed the province has one of the greatest concentrations of ultramafic rocks in North America.

The Coast Ranges of northern California and southwestern Oregon are mostly composed of graywacke and shale of late Jurassic and Cretaceous age. An outlier of Klamath Mountains rock occurs in the southwestern Oregon Coast Ranges, however, and in California a narrow band of metamorphic rocks, the South Fork Mountain schist, lies along the boundary between the Coast Ranges on the west and the Klamath Mountains and Great Valley provinces on the east.

Most of the graywacke and shale of the Coast Ranges is part of the Franciscan Formation in California (Bailey and others 1964) and its probable correlatives in Oregon, the Otter Point and Dothan Formations (Dott 1971). The Franciscan and related formations also include volcanic rocks, radiolarian chert, serpentinite, and occasional tectonic blocks, or "knockers," of exotic schist. Soil development on these various geologic formations depends to a considerable extent on rock weathering. For example, soils throughout the Klamath Mountains and Coast Ranges of northern California and Oregon that developed from mafic volcanic rocks tend to be redder, stonier, lighter textured, and thinner than soils that developed from adjacent sedimentary rocks. And soils derived from quartz-mica schist in southwestern Oregon are deeper, redder, and support

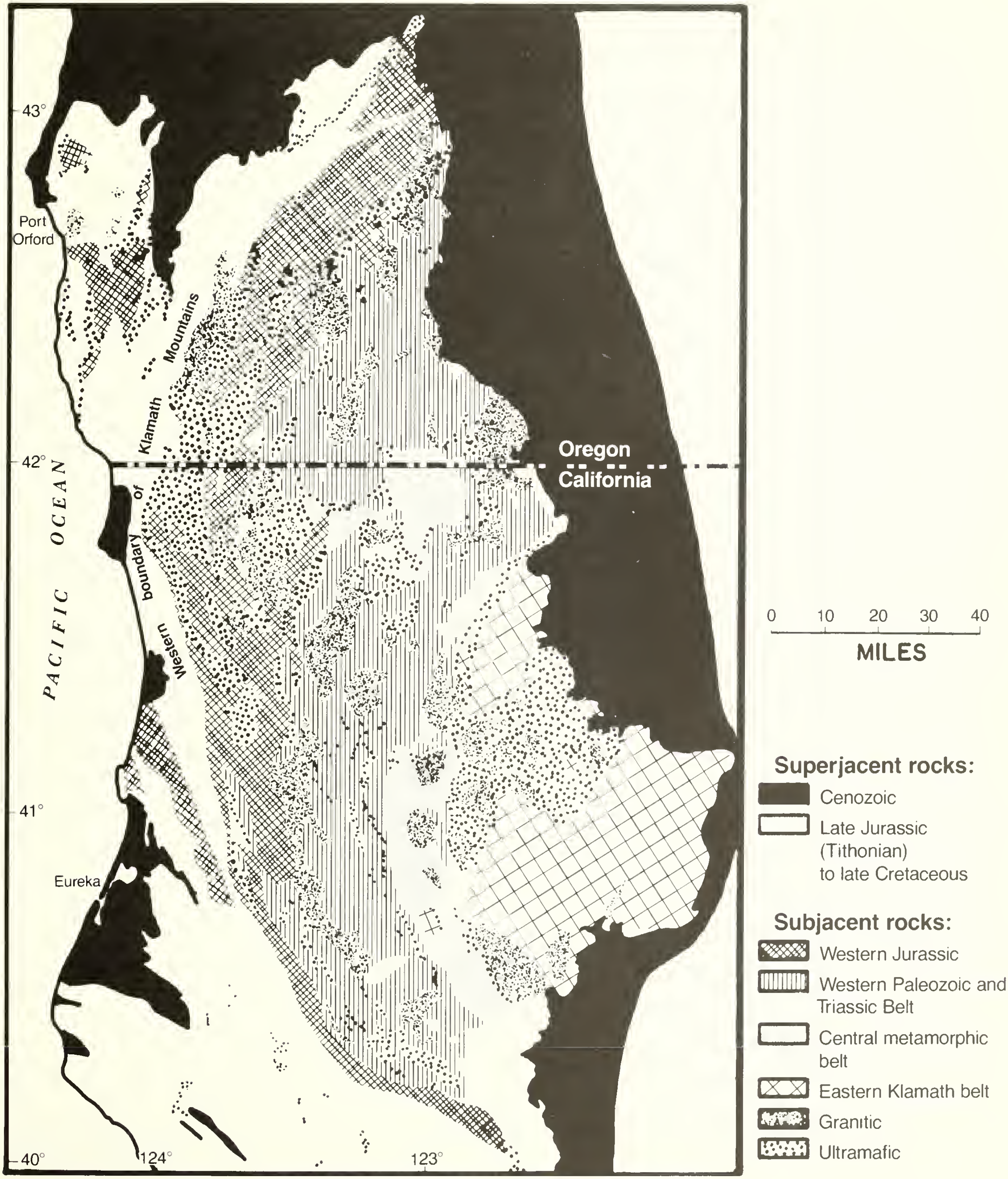

Figure 2—Superjacent and subjacent rocks characterize the geology of the Klamath Mountains Region of southwestern Oregon and northeastern California.

greater timber production than soils derived from adjacent graywacke sandstone (Buzzard and Bowlsby 1970).

In sequence of seven coastal terraces in southwestern Oregon, the sediments on the two highest terraces contain lower Pleistocene molluscan fossils and have weathered to strongly developed Ultisols (Janda 1971). In contrast, lower terraces bear only Spodosols and Inceptisols. On some of the flatter portions of the higher terraces, Spodosols have been superposed on the upper portions of older Ultisols. These observations suggest that the difference in profile characteristics between higher and lower terraces is the result not only of different durations of weathering but also of different weathering environments (Janda 1979).

In the Klamath Mountains province, the parent material consists chiefly of acid igneous and metamorphic rocks. A typical topographic sequence of soils series on schist rock in northwestern California is as follows (Zinke and Colwell 1965): At the lower elevations on the steep slopes in the Trinity River canyon, the sequence begins with the less developed Sheetiron soil series. It then proceeds upslope to the more developed Masterson and Orick soils on the less steep slopes, and finally to the well-developed Sites soils series on the nearly flat ridgetop. This sequence illustrates a pattern in which the most developed soil often is found on plateau-like ridgetops and always on apparently older surfaces in this rough terrain.

The reverse pattern is found along a transect in the California Coast Ranges province (Zinke and Colwell 1965). On a long slope of increasing elevation, a sequence of soil profiles of lessening degree of development with increasing elevation is located on graywacke sandstone. The transect begins with the better developed Josephine soil series at the lower elevations. With increasing elevation it progresses through the lesser developed Hugo soil series to the least developed Hoover series. Similar sequences are found on granitic and andesitic parent materials.

Douglas-fir does not grow well on serpentinite soils developed from ultrabasic parent materials. These soils are characteristically infertile. They are low in nitrogen, phosphorus, and potassium; low in exchangeable calcium; and high in exchangeable magnesium.

Climate

The climate of the Klamath Mountains and Coast Ranges of northern California and southwestern Oregon can be characterized broadly as mild, with wet winters and dry summers (Elford 1970, Sternes 1967). The duration of the wet and dry seasons, as well as the total yearly precipitation, varies with latitude and is influenced particularly by local and regional topography. Most of the Douglas-fir in the region grows within the annual precipitation range of 40 to 100 inches (102 to 254 cm).

The principal atmospheric factors controlling the climate of the region are the Pacific High pressure cell and, to a lesser extent, the Aleutian Low. The Pacific High, so named for its semipermanent location over the northeastern Pacific Ocean, is one of the major eddies in the earth's atmospheric circulation (Shumway 1979). In response to seasonal fluctuations in solar energy, this clockwise-rotating mountain of air migrates northward during late spring and early summer, and then slowly recedes southward during fall and winter. Conditions near the Aleutian Islands favor generation or intensification or both of somewhat smaller eddies that rotate counterclockwise. These low pressure areas, once set in motion, usually drift eastward and are the moisture-bearing storms that move onshore along the Pacific Coast with maximum frequency during winter.

Although somewhat fewer lows are generated during summer, the position of the Pacific High dictates the frequency of storms that penetrate the Klamath Mountains region. During late spring, the northward extension of this vast mountain of air effectively blocks and diverts storms generated in the Aleutians to higher latitudes. After reaching maximum development in July and August, a gradual deterioration and southerly recession of the revolving air mountain allows a steadily increasing number of Aleutian-born storms to batter the Pacific Coast at lower and lower latitudes. The coincidence of maximum eddy development in the Aleutians with maximum southward retreat of the Pacific High causes unusually wet winters along the west coast of North America, between latitudes 40° N and 60° N. Similarly, minimum eddy development with maximum northward protrusion of high pressure causes summers in the Klamath Mountains Region to be remarkably dry.

Between the leeward slopes of the coastal mountains and the foothills of the Cascade-Sierra Nevada ranges is a low, relatively narrow corridor, extending from northern California through Oregon and Washington to the northeast of Vancouver Island. In this area, annual precipitation is less than 1.3 m (50 inches). The area is shielded from heavier precipitation to the west by the coastal mountains and is often short-changed or even bypassed by precipitating systems moving in from the Pacific (Shumway 1979).

In a manner somewhat analogous to regulation of seasonal distribution of precipitation by the Pacific High, the temperature over the Klamath Mountains Region is dominantly influenced by the Rocky Mountains, and to an even greater extent by the Cascade-Sierra Nevada ranges (Shumway 1979). These mountain systems form two separate, substantial barriers to invasion of air masses from the east—air masses that are much colder in winter and considerably warmer in summer.

Normally, these mountain ranges hold back extreme types of weather, which develop in the continental interior, thus allowing the Pacific Coast to remain under the moderating influence of the Pacific Ocean. Water temperature offshore remains relatively constant throughout the year—around 50° F (10° C)—so that any air mass entering unobstructed from the west is warmed in winter and cooled in summer (Shumway 1979). The maritime influence extends inland across the coastal mountains in parts of Oregon and Washington, but in northern California it is limited to a

narrow corridor west of the Coast Range. Considerably more severe continental climates prevail a relatively short distance inland.

Vegetation

Two vegetation zones are prominent in the Klamath Mountains Region; the Mixed Conifer Zone and the Mixed Evergreen Zone (Franklin 1979). Characteristic species in the Mixed Conifer Zone include Douglas-fir, incense-cedar, sugar pine, white fir, and ponderosa pine.[1] Of these species, Douglas-fir and incense-cedar are the most important in terms of timber volume.

Succession is a slower process in the Mixed Conifer Zone than in the Western Hemlock Zone, which is regarded as representing modal or average conditions of temperature and moisture for the Douglas-fir Region as a whole (Franklin 1979). Regeneration of trees in the Mixed Conifer Zone generally is slowed by the more severe environment. In addition, brushfields dominated by chaparral-like evergreen shrubs frequently develop. Typical species are varnishleaf ceanothus, mountain whitethorn ceanothus, narrow-leaved buckbrush, golden chinkapin, canyon live oak, and hairy manzanita. Brushfields slow the rate of establishment and growth of conifers.

The Mixed Evergreen Zone is located in the western Siskiyou and Klamath Mountains (Franklin 1979). Average sites typically are occupied by an overstory of mixed conifers, with sclerophyllous broadleaved trees in the understory. The most important forest trees, which are probably major climax species, are Douglas-fir and tanoak. A variety of other trees also may be present, the most characteristic being hardwoods such as Pacific madrone and golden chinkapin. Conifers that may be present include sugar pine, ponderosa pine, incense-cedar, Jeffrey pine on serpentine, Port-Orford-cedar on moist sites, and knobcone pine on recent burns, Understory communities (low shrub and herb layers) typically are not well developed. Poison oak, Oregon grape, California honeysuckle, oceanspray, California hazel, and rose are characteristic shrubs in the Illinois River region and indicate a more xeric environment than typical of the Western Hemlock Zone. Succession is often slow in the Mixed Evergreen Zone, at least as to development of new conifer forests after disturbance. Sites can be lost easily to evergreen hardwood stands or to dense evergreen chaparral.

In northwestern California, the Douglas-fir forest type develops best on northerly aspects, deeper soils, and the less steep slopes. Generally, Douglas-fir is much less successful on southerly aspects (Hopkins 1964). On poorer sites—particularly ridgetops—and on occasional flats, sugar pine, ponderosa pine, and incense-cedar may predominate; and on the southern and eastern fringes of the region, ponderosa pine and sugar pine begin to replace Douglas-fir (Hopkins 1964). Hardwood understories are heavier nearer the coast (Hopkins 1964). The optimum climate for hardwood species that compete with Douglas-fir in California is found in the extreme northwest corner of the State—an area approximately coincident with Del Norte County, where hardwoods form dense growth, even under heavy overhead canopies.

The present floristic pattern in the California portion of the Klamath Mountains Region reflects a comingling of the Pacific Northwest and California-centered species (Sawyer and Thornburgh 1977). Existing vegetation patterns are controlled primarily by differences in parent material, and secondarily by elevation and by soil moisture as it relates to topography (Waring 1969, Wittaker 1960). Klamath montane forests grow mostly above low-elevation coniferous forests. And dominant species such as Douglas-fir, ponderosa pine, and sugar pine are typical of low as well as montane elevations.

Two subregions—western and eastern—have been identified within the Klamath Mountains Region (Sawyer and Thornburgh 1977). In the western subregion, low-elevation Douglas-fir/hardwood forests give way in the higher elevations to Douglas-fir/white-fir, then to white-fir/California red fir, and finally to California red fir/mountain hemlock. These forests grow in relatively cooler and moister climatic areas than those in the eastern subregion. They are subject to greater maritime influence because of their location (e.g., the Siskiyou Mountains), or because of an overall westerly aspect (e.g., the Marble Mountains, western Salmon Mountains, and western Trinity Alps). In the eastern subregion, the climate is drier and more continental. The forest zonation pattern is comprised of dominantly ponderosa pine forests at low elevations, giving way at higher elevations to pine/white fir, then to white fir/California red fir, and finally to California red fir/mountain hemlock.

Within each of the elevational zones in the Klamath Region, forests along streams have an abundance of herbs and shrubs. In contrast, floristic diversity is lacking on dry slopes, and understory vegetation is virtually absent. With change in elevation, each species assumes a different ecological role by virtue of its response to altered environmental conditions. The resulting pattern is one of changing forest composition as elevation increases. In the western subregion of the White Fir Zone (the lowest of four elevational zones), white fir is the most competitive species in all habitats. Therefore, this species, along with Douglas-fir, dominates mature forests. In the drier eastern subregion, Douglas-fir plays a less competitive role, and other species such as ponderosa pine are less environmentally restricted. The resulting mature forests are of more mixed character than those on comparable habitats in the western subregion.

Trees that are regionally common have broad habitat requirements, high competitive abilities, or high colonizing abilities (Sawyer and Thornburgh 1977). Thus, in the western subregion of the White Fir Zone, Douglas-fir can main-

[1]Scientific names of plants and animals are listed in the *appendix*.

tain significant stand importance in competition with white fir because of its wide ecological tolerance, longevity, and high colonizing ability. With increasing elevation, however, Douglas-fir becomes progressively less competitive. In the higher-elevation zones, the species is confined mainly to xeric habitats.

SEED PRODUCTION

Because Douglas-fir does not reproduce by sprouting, the quantity, quality, and frequency of seed crops are of prime importance for both natural and artificial regeneration. Douglas-fir trees seldom produce seed before they are 10 years old and 15 feet (4.5 m) tall (Isaac 1943). Open-grown trees begin producing seed in appreciable amounts at age 20 to 30 years, but do not produce maximum crops until they are 200 to 300 years old (Fowells 1965). On the average, heavy seed crops are produced every 5 to 7 years in the Pacific Northwest (Isaac 1949). Between heavy seed crops, at least one crop usually fails, and two or more crops are light or medium (Fowells 1965).

Seed Quality

Seed crops in old-growth stands of Douglas-fir in northwestern California may be scanty for several years, and are generally of poor quality (Roy 1960a). Only 18 percent of the total seeds collected over an 8-year period were sound. Seed quality may be much better in young stands. Seed viability was 53 percent in the unthinned portion of a 49-year-old stand in western Washington (Reukema 1961).

The poor quality of the seed from old-growth trees was attributed only partly to insect damage, and mainly to the lack of embryo and endosperm in many seeds with fully formed seed coats (Roy 1960a). The problem appears to originate from either a lack of pollination or fertilization. Parthenocarpy—the development of unpollinated cones with full-sized but empty seed coats—is common in many species, including Douglas-fir (Allen 1942). A small percentage of Douglas-fir seed from parthenocarpic cones may be viable and able to germinate and produce seedlings (Allen 1942, Orr-Ewing 1957).

Cone Crop

The number of reproductive bud primordia initiated would appear to be a promising indicator of a good cone crop, but not all of them develop. Some primordia abort early and disappear; others develop bud scales, then cease further development and become latent (Owens 1969). Thus, the number of cones produced is not determined directly by the number of primordia initiated, but by the proportion of primordia that develop as reproductive buds. Abundance of maturing cones on shoots formed the previous year greatly reduces the possibility of these primordia becoming reproductive buds.

Rain during the pollination period appears to have no major adverse effect on seed set (Silen and Krueger 1962). Cones receptive to pollen during predominately rainy periods showed little reduction in filled seeds when compared with cones receptive during clear weather.

Shade, on the other hand, does affect the formation of flower buds. A month of shade during shoot elongation definitely reduced female buds and increased male buds for a year, but a year later, both male and female bud numbers were reduced (U.S. Dep. Agric., Forest Serv. 1971). This finding suggests that the start of flowering, or at least some necessary precondition for flowering, occurs 26 to 30 months before seedfall.

The influence of shade in reducing the number of flower buds formed probably also accounts for the distribution of cones within the crowns of individual trees. Among 20 open-grown Douglas-fir trees between 10 and 39 years of age, cone numbers were greatest near the tips of branches in the upper two-thirds of the crown on the south-facing side of the tree (Winjum and Johnson 1964). Cones also were longest, and cut-counts (the number of seeds exposed by cutting cones longitudinally) were greatest, in cones from these parts of the crown.

Stimulating Production

Attempts to stimulate seed production by Douglas-fir have not been entirely successful. Techniques most frequently tried include fertilization, thinning, and girdling. On Brush Mountain of the Six Rivers National Forest, northwestern California, seed production did not increase significantly after mature Doubls-fir were fertilized with nitrogen and phosphorus. Urea (43 percent active) provided nitrogen at 200, 400, and 600 pounds per acre (224, 448, and 672 kg per ha), and super triple phosphate (52 percent active) provided P_2O_5 at 2000 pounds per acre (224 kg per ha). Studies in a mature stand of Douglas-fir on Vancouver Island likewise showed no noticeable response in seed production during a 3-year period after fertilization (Crossin and others 1966).

Fertilization of younger stands to stimulate seed production has had variable success. In a 35-year-old stand of Douglas-fir in Washington, ammonium nitrate was applied at rates of 200, 400, and 600 pounds per acre (224, 448, and 672 kg per ha). None of the plots (including unfertilized) had any appreciable cone crops in the first, second, or

fourth years after fertilization. The fertilized plots did, however, produce more cones the third year, although the magnitude of the increase was not reported (Reukema 1968).

More encouraging results were reported for plots treated with 200 pounds per acre (224 kg per ha) each of nitrogen and phosphorus, in the Pacific Northwest. The treated plots produced more than eight times as much seed as did unfertilized plots (Youngberg 1968). A similar eightfold increase in seed production in a thrifty 20-year-old stand of Douglas-fir in southwestern Washington was produced after 200 pounds per acre (224 kg per ha) each of nitrogen and phosphorus were applied (Steinbrenner and others 1960).

The form in which nitrogen is supplied may determine whether cone production is stimulated. Increasing rates of nitrogen, applied as nitrate to 13-year-old Douglas-fir trees at time of vegetative bud-break, increased cone production the following year from two to seven times, whereas ammonium nitrogen produced no response (Ebell and McMullan 1970). Shoot growth, however, was similar for the two forms of nitrogen. Nitrate fertilizer increased the number of reproductive bud primordia.

Thinning, under certain conditions, also stimulates seed production. Three thinning treatments are being tested in a 49-year-old stand of Douglas-fir in western Washington. Although seed production among the thinning regimes did not differ significantly, each thinned plot produced about twice as much seed in a good year as unthinned plots (Reukema 1961). Thinning did not alter the proportion of sound seed produced, nor did it stimulate seed production in poor seed years.

Stem girdling and other forms of stem injury also have been reported to stimulate cone production. In British Columbia several double-stemmed Douglas-fir trees were used to study cone production response to partial girdling. On each of three such trees, one stem was girdled and the other not. Two of these trees were girdled in August 1957, and the third was girdled in May 1958. In fall 1959, cone production from the three girdled stems averaged 7.4 times more than that of their ungirdled partners, but declined to 1.6 and 2.3 times that of the control stems in the next 2 years (Ebell 1971). In the same study, single-stemmed trees were girdled at weekly intervals. The onset of flowering was found to be the best time to apply this technique. Responses obtained by girdling between time of flowering and vegetative bud-break were variable, and cone production was adversely affected when trees were girdled later than 1 week after vegetative bud-break.

Cone production reduced carbohydrate concentration in shoots of all ages, as well as the growth and number of new shoots, and the number of developed buds per shoot (Ebell 1971). These findings help to explain the absence of consecutive abundant cone crops and suggest that cone-inducing treatments should not be applied in good flowering years, because they would be out of phase with the tree's ability to respond to such stimuli.

Losses to Insects

At times, insects cause serious damage to Douglas-fir cones and seeds. Although more than 60 species of insects have been found in Douglas-fir cones (Keen 1958), only 4 regularly cause significant damage in California (Koerber 1960): cone moths *Barbara colfaxiana* (Kearf.) and *Dioryctria abietella* (Dennis and Schiffermüller), seed chalcid *Megastigmus spermotrophus* (Wachtl), and cone midge *Contarinia oregonensis* (Foote). These species as well as another cone midge, *C. washingtonensis* (Johnson), are also the principal insects that damage Douglas-fir cones and seeds in Oregon (Lavender 1978).

The proportion of the seed crop destroyed by insects varies greatly from year to year. When the seed crop is large, losses to insects may be only 3 to 5 percent, while insects may destroy nearly all of a small crop. At three locations in Humboldt County, California, the proportion of Douglas-fir seed destroyed by insects ranged from 46 to 95 percent in 1959 and from 26 to 63 percent in 1961 (Koerber 1962, 1963).

SITE PREPARATION

Successful regeneration of a logged area, regardless of method, usually requires site preparation. Although occasional examples can be found of well-stocked young-growth stands of Douglas-fir that developed after logging without special site preparation or, for that matter, without a conscious effort at regeneration, these are fortuitous exceptions to the general rule. In the Klamath Mountains Region, prompt regeneration of Douglas-fir in areas logged or burned requires direct seeding or planting.

Site preparation generally has three main objectives: removing or consolidating debris left after logging, thereby reducing impediments to planting or seeding and future management operations; killing or substantially weakening shrubs, hardwoods, grasses, and other vegetation that might compete seriously with conifer seedlings; and preparing a mineral soil seedbed. Fulfilling the first objective also reduces the fire hazard—an important consideration for future protection of the new stand. Additional objectives (Stewart 1978) are reducing compaction or improving drainage of surface and upper soil horizons, creating a more favorable microsite on harsh sites, and controlling diseases.

Intensive site preparation is essential to ensure successful establishment of a new crop of trees (Buck 1959). At least 70 percent of the area on a cutblock should be bare mineral soil. Establishment and early growth of Douglas-fir is proportional to the original amount of bare area, regardless of whether regeneration is by natural seeding, artificial seeding, or planting.

The four most common site preparation methods are mechanical preparation, prescribed burning, chemical treatment, or combinations of these (Cleary and others 1978, Stewart 1978). In deciding which method is best in a given situation, several factors must be considered, including slope steepness, soil characteristics, the amount and distribution of slash, the density and probable response of the competing vegetation, and the relative costs of the methods. In some instances, adverse effects on soil and water may preclude use of a particular method in a given locality, or impose restrictions on how the method may be used.

Mechanical Techniques

Mechanical site preparation includes a wide range of techniques, from hand scalping of individual seeding or planting spots, to scarification as a byproduct of logging, to thorough removal of brush and debris by large tractors with various types of blades or other attachments. Generally, the degree of site preparation achieved solely through logging disturbance is highly variable, probably because it is only a byproduct of logging and usually is not included as an objective in the logging plan (Stewart 1978). Also, many shrubs resprout from crowns and roots after logging injury, and thus reliance on logging disturbance alone is rarely successful as a means of suppressing competing vegetation.

Not only is logging disturbance often inadequate site preparation, but it may have varying degrees of adverse effects on the site. In the Oregon Cascades, the effect of four logging methods—tractor, high-lead, skyline, balloon—upon soil surface conditions was studied (Dyrness 1965, 1967, 1972). Of particular interest are the percentages of the surface which were classified as either "compacted" or "deeply disturbed". In order of decreasing percentages of areas in these two condition classes combined, the logging methods ranked as follows: tractor (35.7 pct), high-lead (18.8 pct), skyline (8.1 pct), and balloon (4.3 pct). Surface-soil bulk densities for compacted areas were significantly higher than for undisturbed areas, indicating decreased soil porosities for the former. Soil compaction hampers seedling root development, both because of the physical impediment and because of reduced aeration in the rooting zone.

Tractor logging on fine-textured soils when soil moisture is high can cause puddling, displacement, and compaction on skidroads. In southwestern Washington, soil compaction on skidroads, which covered about 26 percent of the area logged by tractors, reduced stocking by nearly 50 percent and the number of established seedlings by two-thirds when compared with off-road conditions (Steinbrenner and Gessell 1956). Seedlings on skidroads were less vigorous and grew less in height during the first 2 years than did seedlings planted elsewhere. Under such circumstances, special site preparation techniques may be warranted to correct soil compaction.

In contrast to the often ineffective site preparation that results from logging disturbance, highly effective site preparation can be achieved by machine clearing after logging is completed. This method, although expensive, can provide maximum exposure of mineral soil, if that is the desired objective. One way to reduce costs is to keep tractors moving forward, working around obstructions rather than attempting complete eradication (Gratkowski and others 1973). Crawler tractors of various sizes equipped with bulldozer blades, rake blades, or toothed brush blades generally are used. Toothed blades can be pushed through the soil to uproot brush species while minimizing movement of topsoil into windrows (Stewart 1978). Formation of windrows leads to loss of topsoil, which may significantly reduce the growth of Douglas-fir seedlings. In one plantation, seedlings on subsoil grew only two-thirds as much as those on topsoil, and in another plantation they grew less than half as much (Youngberg 1979).

Advantages of mechanical site preparation techniques are these:

- Effective on sites not suited to prescribed burning or on sites with dense stands of brush species that are resistant to herbicides (Stewart 1978).
- May reduce planting costs due to the removal of live or standing dead brush.

Disadvantages are these:

- High operational costs.
- Not practical on slopes steeper than 30 percent, because of possible hazard to equipment operator, and—on certain soil types—may induce accelerated erosion.
- Most effective during summer, when soil compaction may be a problem.
- Loosens topsoil thereby increasing erosion hazard and germination of dormant brush seeds buried in the soil (Gratkowski and others 1973).

Prescribed Burning

Burning alone also can prepare sites satisfactorily under certain conditions but usually entails some element of risk, because the kind of weather and fuel moisture conditions that produce clean burns also make control of fires difficult. For maximum effectiveness, all hardwoods remaining on a cutblock should be felled before burning (Buck 1959). Recent advances in ignition technology, notably the use of a helicopter to achieve ignition with great speed and accuracy throughout a cutblock, have permitted much better control of burning patterns and fire behavior, and greater safety for the slash-burning crew.

Although in many instances burning has been the only feasible way to reduce the large accumulation of logging debris left on a cut-block, studies have shown that burning also produces some undesirable effects. Most of these detrimental effects are associated with "severe" burning—intense, prolonged fire that consumes all duff and changes the color of the upper layers of mineral soil, usually to red (Morris 1958). Fortunately, studies have shown that in normal slash disposal operations, less than 5 or 6 percent of

the total area of a cutblock is subjected to severe burning conditions (Morris 1958, Tarrant 1956).

The detrimental effects of severe burning include these: a reduction in the soil macroscopic pore space and percolation rate, with a resultant increase in runoff and erosion (Tarrant 1956); creation of a crusty, compacted surface; an increase in soil surface temperatures (Neal and others 1965); and a long-term decrease in total nitrogen (Neal and others 1965, Knight 1966), the higher the temperature of the burn, the greater the nitrogen loss (Knight 1966). On steep slopes where the retention of vegetation and duff contributes to soil stability, burning can destroy these protective elements, resulting in increased soil movement (Mersereau and Dyrness 1972). Conversely, burning stimulates the germination of seeds of certain brush species, notably those in the genus *Ceanothus* (Gratkowski 1961c), which often results in serious competition to young conifer seedlings a few years later.

The detrimental effects of burning on conifer regeneration appear to be more serious on some soil types than on others. The regeneration success on burned and unburned cutblocks situated on erodible granitic soils of the Klamath National Forest in northwestern California was evaluated (Heavilin 1977). The cutblocks had been hand-planted 6 years earlier with 2-0 Douglas-fir seedlings. Conifers were seven times more numerous on the unburned than on the burned cutblocks, and the average height of the tallest conifer on each sample plot was nearly twice as great on the unburned as on the burned cutblocks. The results suggest that modification of slash-burning policies on this soil type may improve regeneration success.

To evaluate the influence of slash burning on regeneration success in the Pacific Northwest, a series of paired plots on cutblocks representing a variety of slopes, aspects, soil types, and slash and brush conditions was established on the Cascade Range of Washington and Oregon, and in the Oregon Coast Range (Morris 1958). Plot sizes ranged from 0.25 to 0.50 acres (0.10 to 0.20 ha). One member of each pair was burned by regular slash-disposal crews while the other was left unburned. For at least 5 years after burning, brush crowns covered more area on the unburned than on the burned plots. In the first season after burning, crowns of live brush covered about 3 percent of the area on the burned plots, and 12 percent on the unburned plots. Species differences also were apparent. The main brush species on unburned plots were vine maple and Pacific rhododendron. On burned plots, snowbrush was most abundant.

Subsequent observations on 13 pairs of these plots on the west slopes of the Cascades near Oakridge, Oregon, showed that only for 5 years after burning was there significantly more brush on unburned than on burned plots (Steen 1966). By 7 years, the differences in brush cover were no longer significant. Similarly, conifer reproduction was significantly more abundant on the unburned than on the burned plots for only the first 6 years after slash burning. Thereafter, differences were no longer significant. Conifers present before logging were not included in these comparisons.

Height growth of Douglas-fir seedlings has not been consistently superior on either burned or unburned cutblocks. Seedlings on unburned cutblocks on granitic soils in northwestern California grew substantially taller than those on burned cutblocks (Heavilin 1977). On the other hand, the opposite was true on the Gifford Pinchot National Forest in southeastern Washington: height growth of 1-year-old seedlings on severely burned soil was significantly greater than that on unburned soil, and 2-year-old seedlings grew best on lightly burned seedbeds (Tarrant and Wright 1955). On yet another area—the H. J. Andrews Experimental Forest in the Oregon Cascades—no significant height differences among seedlings growing on unburned, lightly burned, or severely burned seedbeds were found. Differences in soil type, vegetative competition, seedling condition and characteristics, and a host of other factors probably account for these apparent anomalies.

The advantages of prescribed burning for site preparation are these:

- Can be used on steep terrain.
- Results in large, easily planted areas.
- Does not cause soil compaction.
- Costs less than mechanical eradication.

Disadvantages include these:

- Fire control can be difficult and expensive.
- Large complement of well-trained personnel is required, limiting the amount of acreage that can be burned.
- Smoke pollution can be a problem.
- Burning is not suitable for highly erodible soils.
- Many shrubs resprout if fires do not kill the roots and root crowns, and fire may induce germination of seeds of some brush species.
- Burning results in loss of soil nutrients—especially nitrogen (Gratkowski and others 1973).

Chemical Treatment

Chemical site preparation exposes no mineral soil but can be effective in retarding competing vegetation. In fact, chemical site preparation alone has only two purposes: to reduce or eliminate competition, and to alter animal habitat (Stewart 1978). Thus, herbicides alone are effective for site preparation only when residual vegetation is highly susceptible, when slash density is low, when litter is light enough to permit seeding, or when brush is sparse enough to allow planting at reasonable cost. Under many conditions, herbicides are used most effectively in conjunction with either mechanical treatment or prescribed burning. The situations where such combinations are most appropriate are described in the next section.

The advantages of using chemicals for site preparation are these:

- It is often the least expensive method of site preparation.
- It requires a minimum of manpower and supervision to treat large areas quickly.

- It produces the least disturbance and does not compact, loosen, or move topsoil, or expose the surface to erosion.
- It can be used on all types of terrain.

The disadvantages of chemical treatment are these:

- Planting can be more expensive amid chemically killed brush.
- Mineral soil necessary for natural or artificial seeding is not exposed.
- Dominant competitors must be susceptible to herbicides.
- Herbicides may be unacceptable near certain areas.
- Sprayed standing brush protects small animals that may damage seedlings from predators.
- Rapid resurgence of vegetation may require early and frequent respraying to assure dominance of the planted trees (Gratkowski and others 1973).

Combination Treatment

Under some circumstances, a combination of mechanical, burning, and chemical treatments is the most effective site preparation. Often herbicides may kill a competing brush species but leave such dense standing dead brush that planting would be difficult if not impossible. The physical impediment must be removed before the area can be regenerated successfully, and mechanical removal is one possible technique. This combination method has an advantage over mechanical clearing alone, in that the herbicide treatment can reduce resprouting (Stewart 1978). The cost of spraying may be more than offset by reduced costs of subsequent mechanical site preparation (Gratkowski and others 1973). Reversing the treatment sequence also provides an effective combination. Sprays applied after scarification can control sprouts and seedlings of brush species, or control invading grasses (Stewart 1978).

The combination of herbicides and prescribed burning also has proved to be an excellent site preparation technique under appropriate conditions. The method has two variations, one termed "brown and burn" and the other "spray and burn" (Stewart 1978). The brown and burn method uses contact herbicides to desiccate leaves and twigs before burning. Because contact herbicides are not translocated into roots, they will not prevent resprouting after burning. Most contact herbicides are highly toxic and must be handled with caution. Therefore, this method is not always advisable.

The spray and burn technique uses translocated herbicides to defoliate and control residual vegetation before burning. Burning is delayed several months to a year or more after spraying to achieve maximum root kill and stem desiccation (Stewart 1978). In northern California, best results are obtained if two growing seasons are allowed to elapse between spraying with phenoxy herbicides and burning (Bentley and others 1971, Green 1970). Even if some resprouting occurs after the spray and burn treatment and necessitates respraying, the resprouts and brush seedlings that occupy the burned area usually are more susceptible to herbicides than full-crowned mature plants (Stewart 1978).

REGENERATION

Natural Regeneration

Factors that affect the success of natural regeneration include a suitable seedbed, the frequency of good seed crops, the quality of seed, and cultural and other practices that can stimulate seed production. Also needed is adequate dispersal distance of seed from parent trees.

Currently, little reliance is placed on natural seeding as a means of regenerating Douglas-fir because of the irregularity of good seed crops, difficulty of timing the timber harvest to coincide with a good seed crop, problem of controlling seed-eating predators, and vagaries of the weather during the critical germination period. Seedlings that develop from naturally disseminated seeds do, however, often increase the conifer stocking on many cutblocks regenerated by one of the other methods.

Seed Dispersal

The amount of seed reaching the ground decreases rapidly as distance from the seed source increases. In studies of seed dispersal in four old-growth Douglas-fir stands in northwestern California, the amount of seed falling 5 to 6 chains (100 to 120 m) from edges of clearcut blocks averaged only one-tenth the amount under uncut timber. Thus, for natural regeneration, cutblock widths should be limited to 8 chains (160 m) (Roy 1960b).

In the wetter climate of western Washington, distance from the uncut timber edge beyond 500 feet (152 m) of a 130-acre (52.6 ha) clearcut block clearly affected stocking. The block was in old-growth Douglas-fir with an understory of younger western hemlock. Five years after logging, 59 percent of the block was stocked with conifers (only 26 percent with Douglas-fir). On nearby small group cuttings that ranged in size from 1.2 to 4.0 acres (0.49 to 1.62 ha), 85 percent of the milacres were stocked with conifers (40 percent with Douglas-fir) (Worthington 1953).

Shade, as well as distance from seed source, can influence the success of natural regeneration. This was apparent in a study of 15 clearcut blocks in the Oregon Cascades, 13 of which were less than 9 acres (3.6 ha) and 2 larger (Franklin 1963). The smaller units were of four types: (1) north-south strip clearcuts, (2) east-west strip clearcuts, (3) rectangular clearcuts, and (4) circular clearcuts. The two larger units were a seed-tree cutting of about 20 acres (8 ha) and a staggered-setting clearcut of about 48 acres (19 ha).

All 13 of the small clearcuttings were regenerated adequately within 4 years after logging, and on 8 of them the number of seedlings increased significantly as the number of hours of shade increased. This relationship was more pronounced on the strip clearcuts oriented east-west than on those oriented north-south.

On the 20-acre (8 ha) seed-tree cutting, 36 seed trees were left on about two-thirds of the area. As on the smaller units, this treatment resulted in adequate regeneration within 4 years after logging. The seed trees had been selected for vigor, and were spaced 120 feet (37 m) apart. The shade cast by these trees helped promote prompt seedling establishment.

The effect of cutblock size on stocking was minor on 13 staggered settings (12 larger than 40 acres) in the Oregon Cascades and Coast Range. In fact, it was less important than were the effects of aspect, ground cover conditions, or burning treatment. Cutblocks having the best stocking had northerly aspects, light slash, light herbaceous cover, and had not been burned (Lavender and others 1956).

Seed Predators

Mice, shrews, voles, chipmunks, and squirrels often reduce or prevent natural regeneration. Foremost among these seed predators is the white-footed deer mouse, which destroys more coniferous tree seed than any other small mammal in western Oregon (Hooven 1958). Deer mice hold a position of similar notoriety in northern California. A caged deer mouse can eat from 250 to 350 Douglas-fir seeds per day (Hooven 1975). At this rate, two deer mice can consume about 40,000 seeds (1 pound) in fewer than 80 days. Trapping for several years after logging on clearcuttings in both the Coast and Cascade Mountain Ranges in Oregon indicated populations of from four to seven deer mice per acre (10 to 17 per hectare).

Small-mammal populations were observed for 11 years in Lane County, west-central Oregon (Hooven 1976). The yearly number of species ranged from 8 to 12 and averaged 10, although 21 species were caught overall during that period. Of these 21 species, 6 comprised 92 percent of all individuals caught. Deer mice accounted for 30 percent of the total, shrews (Trowbridge's shrew and vagrant shrew) 25 percent, Townsend's chipmunks 19 percent, Oregon creeping voles 12 percent, and jumping mice 6 percent.

During the 11-year study, high populations of small mammals generally were related directly to moderate-to-good cone crops of Douglas-fir the previous year. Although the overall number of small mammals increased remarkably, only deer mice and shrews showed a definite, constant increase during any year immediately after a moderate-to-good cone crop. The densities of the other seed-eaters fluctuated, and often did not correspond to an increased availability of seed.

In another western Oregon study, mice (mostly deer mice) and shrews destroyed 41 percent of the naturally disseminated Douglas-fir seed between seed fall and the end of germination the following year (Gashwiler 1970). Birds (predominantly) and chipmunks destroyed another 24 percent, mostly before germination began.

A hot slash burn can kill or drive out all resident mice, but they reinvade within 2 1/2 weeks (Tevis 1956a). However, in a forested area devastated by an intense wildfire that burned about 87,000 acres (35,200 ha) northwest of Yreka, California, mice and kangaroo rats survived the conflagration by burrowing underground, and then fed on insects, or on seeds in hidden stores (Tevis 1956b).

Shrews may consume sizable amounts of Douglas-fir seed; but seeds are not their primary food, and they cannot survive on seeds alone. On the Tillamook Burn in Oregon, shrews consumed the greatest number of Douglas-fir seeds in winter and early spring, when their chief natural foods—insects—were scarce (Kangur 1954). Chipmunks and squirrels sometimes are important seed predators locally, but their overall impact is generally less than that of mice and shrews. Varied thrushes, spotted towhees, mountain quail, golden-crowned sparrows, fox sparrows, and especially juncos all benefit from logging and consume most of the conifer seeds lost on the ground to birds (Hagar 1960).

Direct Seeding

Direct seeding plays only a minor role in current planning for Douglas-fir regeneration. Although not all of the drawbacks of natural seeding apply to direct seeding, adequate protection of the seeds from seed-eating rodents still poses a problem, and unfavorable weather patterns can result in germination failures or early mortality of newly germinated seedlings.

Direct seeding of Douglas-fir is still of interest to some forest managers because large areas can be seeded quickly with a helicopter, and because the initial cost is considerably cheaper than planting. However, the probability of a seeding failure is comparatively high, even under ideal conditions on maritime slopes. If an area must be reseeded one or more times, the cost advantage over planting is lost. The difficulty lies not so much in a lack of knowledge of the requirements for successful seeding, but rather in a lack of control over some key factors that determine success or failure.

Condition of Seedbed

One important factor over which the silviculturist has considerable control is the condition of the seedbed. Seed germination and seedling survival of Douglas-fir were studied on six seedbed types on a south-facing clearcut in the Oregon Coast Range (Hermann and Chilcote 1965). Seedbeds studied were sawdust, litter, charcoal, and soils that were unburned, lightly burned, or severly burned. Each seedbed was exposed to three light intensities—100, 75, and 25 percent of full sunlight. Regardless of light conditions, germination was best on charcoal and on severely burned soil. Good germination was attributed to prolonged preservation of moisture at the surface of these seedbeds. Germi-

nation differed significantly from that on the other four seedbeds, with better stocking maintained for at least six growing seasons.

Wood ash, on the other hand, may inhibit seed germination by reducing the oxygen supply to seeds, which become buried in the ash (Tarrant 1954). High alkalinity of the wood ash did not inhibit germination, but seedlings suffered heavy damping-off losses a day or two after emergence—a common occurrence on seedbeds with high pH.

Soil type is another factor in successful Douglas-fir regeneration. In northwestern California, more seedlings became established on fine-textured soils that were predominately reddish-brown than on coarser-textured gray-brown soils (Strothmann 1971b). The specific soil properties responsible for this difference were not identified, but doubtless the greater water-holding capacity of the fine-textured soil was one such property.

Time of Seeding

Timing of direct seeding can contribute to successful regeneration. Seeding in late fall or early winter appears to yield better results than that in late winter or spring (Dick 1962, Lavender 1958a, Strothmann 1971b). Late fall or early winter coincides with natural seed fall, and allows ample time for the chilling required to break seed dormancy.

Spring seeding of stratified (artificially moist-chilled) seed has been suggested as a way to reduce the time that seeds lie on the ground exposed to rodents, birds, and other predators before conditions suitable for germination occur. Seeding in November and December produced the best results, regardless of whether the seeds were stratified (Lavender 1958a). Stratification increased the speed of germination, but not the total amount.

Site Aspect and Temperature

Northerly aspects apparently favor seeding success in most instances (Steele 1953, Lavender 1958b), but some exceptions exist. On fine-textured soils in northwestern California, more seedlings survived and grew on south aspects than on north, although the reverse was true on coarser-textured soils (Strothmann 1971b). Seeds probably germinate sooner on south slopes because soils there warm earlier. This initial advantage on south slopes may persist through the subsequent growing season on fine-textured soils because of slower soil moisture depletion than on coarse-textured soils.

On south aspects, temperature at the soil surface may exceed 150° F (66° C) (Strothmann 1972). Under laboratory conditions, 2- to 12-week-old Douglas-fir seedlings die after 4 hours of exposure at 122° F (50° C) (Silen 1960). Further, the seedbed material, temperature, and duration of exposure interact to affect mortality. On eight clearcut blocks in western Oregon, the melting of a 138° F (59° C) temperature pellet was most closely related to mortality, although the lethal temperature threshold ranged from 125° to over 150° F (52° to 66° C).

In many instances, physiological drought rather than soil surface temperature, per se, may be the actual cause of seedling mortality. In northern California we have observed over the course of many years hundreds of young Douglas-fir seedlings within a few days of death and found no positive evidence—stem lesions at ground line—that mortality was caused by heat damage. Although soil moisture may be available, young seedlings apparently do not have root systems adequately developed to provide the water needed to counter transpirational losses during periods of hot, sunny weather. This condition, coupled with poor foliar control of transpiration in developing foliage, causes seedlings to desiccate and die.

One solution is to provide shade for seedlings on severe sites. On the Hoopa Indian Reservation in northwestern California, a 50 percent shade treatment with a lath framework produced the highest germination and 2-year survival of Douglas-fir seeded on a hot, south-facing cutblock (Strothmann 1972). Plots were not visited frequently enough to pinpoint the causes of mortality.

On production seeding jobs, effective shade can be provided by sowing a nurse crop, such as brown mustard, after slash disposal—a technique used successfully on south slopes in Oregon (Chilcote 1957). The mustard (or any nurse crop) must, however, be sprayed with herbicide the following spring to convert it to dead shade and eliminate its competition for soil moisture after the rainy season ends. Other potentially useful but untested nurse crop species are New Zealand fireweed and Australian fireweed, which are annuals and seed naturally.

Causes of Loss or Damage

Determining the causes of loss or damage to seeds and seedlings is often difficult enough, and unless the site is visited frequently during the critical period of germination and establishment, pinpointing these causes may be impossible. Even before they reach the ground, seeds may be damaged by the seed dispersal mechanism of the seeding aircraft. Germination of a given lot of endrin-treated Douglas-fir seed was significantly lower when disseminated by one helicopter than when disseminated by another (Edgren 1968).

Once seeds reach the ground, numerous causes of loss come into play. A detailed study of February-sown Douglas-fir seed in western Washington showed that only 28 percent germinated, and that almost half of the seeds were destroyed before germination (Lawrence and Rediske 1962). Pregermination losses amounting to 45 percent were attributed to molds (19 pct), insects (11 pct), rodents (8 pct), birds (3 pct), and mechanical injury and unknown (4 pct). Another 27 percent, comprised of whole seeds that showed no evidence of external damage, simply failed to germinate. Postgermination losses totalling 15 percent were caused mainly by damping-off fungi (9 pct). Thus only 13 percent of the sown seeds germinated and became seedlings that survived a full year in the field.

Young seedlings can be damaged in numerous ways. In northern California, local erosion and deposition were the major causes of seedling loss on soils derived from decomposed granite (Roy 1961). Other causes of mortality were drought (10 pct), deer browsing (2 pct), and sunscald or heat injury (1 pct). After two growing seasons, 31 percent of the survivors had been injured by deer browsing and 13 percent by frost.

Seeding Methods and Seed Protection

Techniques for protecting seeds against predators vary, depending upon the seeding method. The spot seeding (or seed spotting) method is the most economical in terms of seed but the most costly in terms of labor. A crew places a few seeds (from 2 to 10) in selected spots throughout the area. The spacing between spots depends largely on expected germination and survival, and on the final stocking density desired.

In spot seeding, seeds usually are covered with a thin layer of soil to improve their chances of germinating. Such covering is inadequate protection against predators. Customarily, seeds are protected either by a coating of repellent-poison, or by conical or dome-shaped hardware-cloth screens.

Besides protecting seeds from rodents and birds, screens also provide a small amount of shade and physical protection from falling debris such as leaves. Screens have serious disadvantages: the high cost of the material; and the costly labor involved in forming it into a conical or hemispherical shape, carrying the screens into the field, installing them, and ultimately removing them to prevent strangulation of the growing seedlings.

To reduce costs, foresters have tried using small, inexpensive berry baskets. They cost only one-fifth as much as conical hardware-cloth screens, and because they also weigh much less, more can be carried into the field on each trip (Zechentmayer 1971). Also, because the baskets are biodegradable, removing them may not be necessary. They share some of the drawbacks, however, with wire-mesh screens. For example, baskets can be crushed or displaced easily by large animals, and frost-heaving of anchor pins can lift the baskets off the ground and render them ineffective (Utterback and Berry 1977, Zechentmayer 1971).

Even though costs associated with spot seeding may sometimes be reduced, the method is slower and more costly than broadcast seeding—especially aerial seeding. For this reason, aerial broadcast seeding has far greater appeal to the forest manager, particularly if large acreages are involved.

With broadcast seeding, as with spot seeding, success is unlikely unless seeds are protected from rodents and birds. Because mechanical barriers are not feasible with this seeding method, the approach has been to use chemicals. In the past, poisoning the area for rodents before seeding was standard practice. The most common materials were thallous sulfate and sodium fluoroacetate (also known as "Compound 1080"). Wheat kernels usually were impregnated with the chemicals and spread over the area several weeks before seeding. Sometimes poisoned wheat was applied again when the conifers were seeded. Usually, a sizable buffer strip was poisoned to prevent reinvasion of the area by rodents.

Although poisoning effectively reduces rodent populations, even in the early days of its use, some of the problems associated with it were recognized: a $^1/_4$-mile buffer strip is required to prevent reinvasion, a two-stage operation is required for seeding, small areas cannot be seeded economically because of the disproportionate cost of the buffer strip, small mammals other than tree seed eaters may be destroyed, and baits deteriorate over winter and do not provide protection in the spring (Hooven 1955).

The phenomenon of bait shyness to protect untreated seed was tested on the Six Rivers National Forest in northwestern California (Tevis 1956c). A deer mouse population that would eat neither Douglas-fir seeds that were poisoned, but sublethal, nor untreated seeds was developed. Douglas-fir seed served as poisoned bait. The results suggested that developing a population of non-seed-eating rodents in a specific area may be easier and more desirable than trying to eliminate large numbers of animals by the standard poisoning operations then in use.

A more widely supported approach to overcoming the disadvantages of earlier large-scale poisoning techniques was to develop a relatively insoluble repellent or toxic chemical that could be applied directly to tree seeds without destroying their ability to germinate. Among the more promising of the earlier chemicals was tetramine (tetramethylene-disulphotetramine). Tests of this chemical in a Douglas-fir seeding trial in western Oregon produced encouraging results (Hooven 1955); but, in northwestern California, an acetone-tetramine treatment both reduced and retarded germination of Douglas-fir seed (Roy 1957).

When tetramine became increasingly more difficult to obtain in the mid-1950's it was superseded by endrin, a chlorinated hydrocarbon. Endrin was tested as a Douglas-fir seed protectant on the Tillamook Burn in Oregon (Hooven 1957). Six months after seeding, the poorest of the endrin-treated plots had more than three times as many stocked milacre quadrats and seven times as many Douglas-fir seedlings as the control plots. And rodent populations sampled by live trapping both before and after seeding were reduced greatly on the areas sown with treated seeds. Other successes in the Pacific Northwest have been reported from broadcast sowing of endrin-treated Douglas-fir seed (Dick and others 1958, Dimock 1957).

Endrin-arasan treated seeds also were tested in northwestern California and were found to be adequately protected against seed-eating rodents (Roy 1961). The endrin-arasan treatment did not inhibit germination. Subsequent to these early studies, the endrin-arasan coating became the standard seed protectant for direct seeding operations throughout the Douglas-fir region. However, use of the endrin-arasan treatment has been reduced significantly as a result of concerns about toxicity and of occasional instan-

ces in which nontarget species have been killed. No satisfactory substitute for endrin has been developed, but research is continuing.

Research also has continued toward developing safer rodenticides for use as area poisons to replace acute toxicants, such as Compound 1080 used earlier. Among the more promising are the anticoagulant rodenticides such as diphacinone. On a Douglas-fir clearcut in northwestern California and in a pine growing area in the Sierra Nevada, consumption of 0.01 percent diphacinone-treated crimped oat groats for a minimum of 3 days was fatal to 80 percent of the deer mice (Howard and others 1970). Longer exposure to the bait frequently produced 100 percent mortality. When 0.01 percent diphacinone bait was broadcast at 2 pounds per acre (2.24 kg/ha) in two field tests, no deer mice tagged before treatment were recaptured.

Future of Direct Seeding

The odds against successfully regenerating an area by direct seeding are formidable, but success is attainable. In November 1960, seven clearcut blocks totaling 276 acres (112 ha) were helicopter-seeded with Douglas-fir on the Hoopa Indian Reservation in northwestern California (Lusher 1964). Seeding was preceded by thorough site preparation, including felling, bunching, and burning of the residual hardwoods after harvesting the conifers. The ground was in excellent condition at the time of seeding, being settled and firm, but not compacted, and favorably moist from recent rains. Seeding was at the rate of 1 pound per acre (1.12 kg/ha).

On four of the cutblocks, seed traps were placed before seeding to monitor seed distribution and quantity of seed delivered. Quantities ranged from about 20,000 to 36,000 seeds per acre (49,000 to 89,000/ha). Milacre plots were established on which the live seedlings on each plot were counted periodically. Milacre stocking at the end of the third growing season was estimated at 43 percent, and number of live seedlings per acre at 1292 (3192/ha). Thus, aerial sowing of endrin-treated Douglas-fir seeds on a fairly large scale can be successful in the Klamath Mountains Region.

Notwithstanding an occasional success, the future of direct seeding of Douglas-fir in the region is uncertain. Aside from the need to develop a safe and effective seed protectant to replace endrin, direct seeding has numerous other drawbacks that have brought about a decline in interest in this regeneration method. Many failures are doubtless caused by unfavorable climatic conditions. The hot, dry summers that characterize the region place a severe stress on newly germinated seedlings, and many succumb to drought during their first summer. In this respect, coastal locations with their more moderate temperatures and higher humidities are more favorable for direct seeding than sites further inland.

Another drawback of direct seeding—particularly aerial seeding—is the lack of control over tree spacing, which results in stocking voids in some places and overly dense stocking in others. Dense stocking often requires precommercial thinning. Furthermore, direct seeding is not suited for efficiently utilizing genetically improved seed developed in tree improvement programs because it requires many seeds to establish a seedling and wastes seeds. For one or more of these reasons—and the stringent stocking requirements of the Forest Practices Acts of California and Oregon, most land managers rely increasingly on planting as the best way to regenerate their forest lands.

Planting

Planting is by far the most common and most reliable method for regenerating Douglas-fir. Although regeneration can still fail, extensive research has improved considerably the chances of successfully establishing a new forest stand by planting. Careful attention to detail during every phase of the planting operation is important, because numerous variables affect planting success: dormancy, time of lifting, root regenerating capacity, stock handling, stock size, age, top/root ratio, planting depth, and type of stock (bare root or containerized).

Most of the Klamath Mountains Region is unsuitable for machine planting because the ground often is steep or rocky or both. Traditionally, bare-root nursery stock has been planted by hand. However, container-grown stock also has been used, especially on forest industry lands. Several of the large industrial landowners have developed nurseries in recent years and currently produce large quantities of containerized seedlings, both for their own use and for sale. However, on large acreages of publicly owned lands—particularly National Forests and Bureau of Land Management lands—bare-root stock is the type most commonly planted. Much of this stock is produced by Forest Service nurseries at McKinleyville, California, and Medford, Oregon, and at the State-owned Phipps Nursery near Elkton, Oregon.

Dormancy, Lifting, Storage

In recent years, much has been learned of the interrelationships among seed source, dormancy, food reserves, root growth capacity, lifting dates, and storage of bare-root seedlings, and how these factors affect ultimate field survival of seedlings. Many of the early planting failures can be attributed directly to ignorance of these interactions and the improper timing of key nursery operations.

In California, Douglas-fir showed a marked seasonal periodicity in root-regenerating potential (RRP) (Stone and others 1962). The RRP declined during summer, rose abruptly during September, was high during winter, and dropped sharply during April. RRP, based on month-long test periods in the greenhouse, was primarily an expression only of lateral root elongation rather than of lateral root initiation and elongation. Therefore, considerable effort can be justified in preventing desiccation and injury of short lateral roots when seedlings are lifted and before planting.

Laboratory and field studies in Oregon showed a similar pattern (Lavender 1964). Seedlings lifted before December, or after buds began to swell in spring, were affected adversely by transplanting. Seedling physiology was disrupted, resulting in reduced growth of shoots and roots, and in poor field survival. The disruption lasted at least through the second growing season after planting.

Food reserves and the seasonal growth of Douglas-fir seedlings were measured at biweekly intervals for more than a year at the Wind River Nursery near Carson, Washington (Krueger and Trappe 1967). Seedlings showed a general pattern of alternating root, diameter, and shoot growth. Rapid root growth did not coincide with rapid shoot elongation, but preceded and followed it. Increased root activity was correlated strongly with lowered concentrations of reducing sugars in seedling roots. Many current nursery practices, such as late fall lifting and lifting before spring bud swelling, are in harmony with underlying physiological events. For example, root growth in the nursery peaks just before or at the time of bud burst. As buds begin to swell, root growth diminishes rapidly, hitting a low point when shoots are flushing rapidly. Root growth remains low during rapid shoot growth, then gradually increases to another peak in July or August after top growth tapers off.

The importance of lifting planting stock only when it is fully dormant increases as planting sites become harsher. Generally, sites in the Douglas-fir Region become harsher (in terms of increased moisture stress to seedlings) towards the south (Hermann and others 1972). Moisture stress also may be induced by vegetational competition. An outplanting test near Corvallis, Oregon, compared survival of 2-0 Douglas-fir seedlings on two sites—one with bare soil, and the other with grass and weeds (Zaerr and Lavender 1972). Seedlings planted on the site free of competing vegetation had good survival, even when lifted late (when actively growing) or stored up to 9 weeks. On the other site, late lifting or storage resulted in increased mortality and adversely affected subsequent weight gain of the seedlings.

Cold storage *per se* does not reduce seedling vigor or ability to survive after outplanting. If seedlings are lifted when fully dormant and stored under proper conditions, field survival can be excellent. Cold storage of seedlings lifted when they are dormant helps to extend the period they are highly capable of root growth (Hermann and others 1972). Nondormant stock, on the other hand, does not store well—even for short periods. Cold storage for periods of 2 weeks or longer adversely affected the vigor of Douglas-fir seedlings that were lifted before December or after bud swelling in spring (Lavender 1964).

Studies at the Humboldt Nursery in McKinleyville, California, demonstrated that dormancy of Douglas-fir is seed-source dependent. The calendar periods during which seedlings from different seed sources can be lifted safely and stored vary (Jenkinson and Nelson 1978). These "lifting windows" are fixed by the seed source response to nursery climate and cold storage. Some sources, for example, can be lifted and stored anytime between early November and early March, and outplanted with first-year survival potentials that range from 88 to 98 percent. Other sources have much narrower windows—in some cases only a 6-week period centered around early February.

The onset of dormancy and the "hardening-off" process for seedlings of all seed sources can be either hastened or delayed by certain nursery management practices—especially the timing of irrigation. At the Humboldt Nursery, for example, hardening-off has been accelerated considerably in recent years by cutting back on irrigation as early as July, in contrast to the earlier practice of continuing irrigation through August. Untimely late summer rains, however, occasionally may nullify these efforts.

Stock Characteristics

Over the years, certain characteristics of bare-root planting stock have been regarded as important determinants of successful field survival. Among these are seedling size, weight, stem diameter, top/root ratio, and the type of root system. Their relative importance may vary under different planting situations.

On shallow rocky soils on south slopes in the eastern foothills of the Oregon Coast Range, the size (shoot length) of Douglas-fir seedlings was not an important factor for first-year survival, provided the seedlings were in good physiological condition when planted (Hermann and Newton 1975). Previous attempts to establish Douglas-fir in the study area had failed. Vegetation consisted of a dense cover of perennial grasses and thimbleberry. Douglas-fir survival was evaluated with and without various site treatments, including irrigation, application of herbicide, shading, and browse-prevention screens. Among these treatments, irrigation was the most effective in improving survival, but was costly and considered not feasible except for special, small-scale projects. In other test plantings in the Oregon Coast Range near Corvallis, first-year mortality was significantly higher for seedlings with fresh weights of less than 4 grams (Zaerr and Lavender 1976).

Evaluation of test plantings on moderate-to-steep, clean, south slopes in the western Cascades, after 4 years showed that transplant stock—both 1-1 and 2-1—survived better and grew taller than 1-0, 2-0, or 3-0 seedlings (Edgren 1977). The differences, however, were not great enough to recommend the exclusive use of comparatively costly transplants over 2-0 seedlings. In these tests, the top/root ratio, by itself, was a poor predictor of survival. Seedlings that survived best (1-1 stock) and poorest (1-0 stock) both had top/root ratios of 1.5 to 1.7—values intermediate in the range tested.

In another test planting in a burned-over area in north-central Washington, survival of 2-0 seedlings with large roots was 22 to 26 percent higher than that of seedlings with small roots (Lopushinsky and Beebe 1976). Increase in shoot mass of large-root seedlings was twice that of small-root seedlings, and height growth of large-root seedlings was 1.2 to 1.7 times greater. Thus, the importance of a good root system was again demonstrated.

On a severe site in the Siskiyou Mountains of southwest Oregon, performance of 2-0 bare-root stock, 1-0 plugs, and plug-1 seedlings—a newer stock type—were compared (Hobbs and Wearstler 1983). After 2 years, survival in the field was proportional to root initiation and growth. Root development and survival (91 pct) were greatest for 1-0 plug seedlings, intermediate for plug-1 bare-root seedlings (87 pct survival), and lowest for 2-0 bare-root stock (56 pct survival). Height and diameter growth did not differ.

Special Treatments

The importance of root condition to planting success has led to numerous techniques for stimulating root growth, developing a bushy root system, or improving the root environment of the outplanted seedling. A common nursery practice is to undercut seedlings in the beds so that bushier root systems develop before lifting. Whether a compact, bushy root system is better than a root system with fewer, but longer, deep-reaching roots for seedling survival is uncertain (Trappe 1971).

The *glauca* variety of Douglas-fir, when root-pruned, initiated clusters of new, long roots immediately above the pruning wounds (Trappe 1971). These new roots tended to grow downward rapidly with no lateral branching. Bushiness, therefore, did not increase. In contrast, seedlings of variety *menziesii* initiated few, if any, new roots. Instead, small rootlets that were dormant began to grow. Many mycorrhizae burst their mantles and grew as characteristic for long roots. Consequently, the number of major branch roots increased in the upper root system, while roots on most seedlings did not grow rapidly downward. To improve planting success, we need to learn more about what type of root system is best, and how such a root system can be produced.

A modification of undercutting called "wrenching" has shown promise for improving the ability of outplanted Douglas-fir seedlings to tolerate drought (Koon and O'Dell 1977). Wrenching differs from regular undercutting in that the cutting blade is set at a slight downward angle, whereas for regular undercutting it is horizontal. Thus, in wrenching, not only are the roots severed, but the entire seedling is lifted slightly and the root zone is aerated.

At the Humboldt Nursery in northwestern California, two undercutting depths (6 and 8 inches [15 and 20 cm]), and two time intervals between wrenchings (2 and 4 weeks) were tried on 2-0 Douglas-fir seedlings (Koon and O'Dell 1977). The 8-inch depth was better. At the end of the first summer after outplanting, mortality was significantly less for seedlings that were undercut to 8 inches at either 2-week (44 pct mortality) or 4-week (47 pct mortality) intervals than for unwrenched seedlings (69 pct mortality). Mortality of seedlings undercut at the 6-inch depth was 56 percent—not significantly less than that for unwrenched seedlings. Wrenched seedlings in all treatments had significantly more reductions in shoot weight, stem diameter, and height than did unwrenched seedlings. Root weight, however, remained largely unaffected. Thus, the principal effect of wrenching was to retard top growth. Wrenching also created a noticeably more fibrous root system.

The timing of undercutting or wrenching helps to determine whether growth of tops or roots is stimulated or retarded and, therefore, whether the desired seedling characteristics are achieved. Performing either undercutting or wrenching at the wrong time can reduce stem diameters or root masses or both (Edgren and others 1978).

At the D. L. Phipps State Forest Nursery at Elkton, Oregon, both wrenched and unwrenched seedlings increased substantially in stem diameter and root dry weight between August and January (Stein 1978). Seedlings were wrenched in early August. Between late August and mid-January, seedling diameters increased about 50 percent. Diameter of unwrenched seedlings tended to be smaller at first, but by January little difference was noted between the wrenched and unwrenched seedlings. Average dry weight of unwrenched seedling roots more than doubled from August to January, but for wrenched seedlings it increased fivefold. In late August (24 days after wrenching), roots of the wrenched seedlings weighed only about half as much as those of the unwrenched seedlings, but by mid-January the weight of wrenched seedlings slightly exceeded that of the unwrenched seedlings. However, neither first-season field survival nor height growth were improved by the early August wrenching.

Besides undercutting or wrenching or both, other special treatments have been tried for stimulating early initiation and growth of roots. Two commercial preparations, Gro-fast and Transplantone, were tested in California on root-pruned 2-0 Douglas-fir seedlings (Osburn 1960).[2] The Gro-fast preparation contained 100 ppm giberellins and was applied in separate treatments to tops, to roots, and to both tops and roots. The Transplantone contained naphthylacetamide and vitamin B_1 and was applied only to roots. Although the study had some serious limitations such as a small sample, small stock in poor condition, and planting in rather small containers, some definite response patterns were observed. Gro-fast, regardless of where applied, did not improve growth over the control seedlings. But Transplantone produced a definite and early stimulation of root growth. Improved length and density of the root system of treated seedlings were maintained throughout the second growing season.

Handling Stock

Many planting failures have been attributed to careless handling of stock at the nursery, in transit, or at the planting site. One form of abuse is exposing seedling roots to excessive drying before planting.

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[2]Trade names and commercial enterprises or products are mentioned solely for information. No endorsement by the U.S. Department of Agriculture is implied.

To ascertain the effects of drying on subsequent survival of seedlings, a series of root-exposure studies was conducted near Corvallis, Oregon (Hermann 1962, 1964, 1967). The last of these studies tested varying periods of cold storage and exposure on 2-0 Douglas-fir seedlings lifted on three different dates. When seedlings were lifted (November 5, January 28, and March 26), none had buds either open or ready to open, though buds were just beginning to swell in March. Seedlings were treated in one of three ways—no storage, 3 weeks cold storage, or 6 weeks cold storage. Cold storage was at 35°F (1.5°C) in closed polyethylene bags. Before planting, roots from each group were exposed for 0, 5, 15, 30, 60, or 120 minutes, in an illuminated chamber where temperature was maintained at 90°F (32° C) and relative humidity at 30 percent. After exposure, some seedlings from each group were potted and placed in a controlled environment chamber. Most, however, were outplanted in plots, which were kept weed-free but not watered.

Exposure of roots delayed bud-bursting, and the longer the exposure, the greater the delay. Storage further delayed bud-bursting. Bud-bursting was delayed least for the seedlings lifted in January. Survival decreased as root exposure lengthened, but was influenced by lifting date and time of storage. Seedlings lifted in January were less sensitive to exposure than were those lifted in November or March. For all lifting dates, seedling survival was affected adversely by longer storage. Another effect of root exposure was a decrease in height growth, needle weight, needle length, and terminal bud length of surviving seedlings. On the average, growth was best for seedlings lifted in January, not stored, and roots not exposed.

Another approach to reducing seedling damage from root exposure is to coat the roots with a moisture-holding material, such as clay slurry, sodium alginate (a seaweed product), or xanthan gum (a hydrophilic colloid of a polysaccharide). In Oregon, roots of 2-0 Douglas-fir seedlings were dipped in one of these materials (and a control group dipped in distilled water), and then exposed for periods of 0, 10, 20; and 40 minutes, at temperatures between 85° and 90°F (29° and 32°C) and relative humidities between 20 and 25 percent (Owston and Stein 1972a). After root exposure, seedlings were potted individually, placed in a greenhouse, and adequately watered for 4 weeks. Effectiveness of the root-coating treatments then was evaluated by measuring seedling water stress.

Only the control seedlings that had been exposed to root desiccation for 40 minutes showed moderate impairment of water uptake after the 4-week recovery period in the greenhouse. Thus, all coatings provided substantial protection against drying from exposure, with xanthan gum considered the best of the three for Douglas-fir. None of these root-coating treatments was recommended, however, if seedlings were to be stored for an extended period. After 8 weeks' storage, followed by 4 weeks of growing in pots in the greenhouse, seedlings with roots that were coated showed significantly higher moisture stress than those whose roots had been dipped in distilled water and then packed in sphagnum moss for storage.

A transpiration retardant (Foli-gard) and a root coating (Rutex 59) were tested on 1-0 Douglas-fir planting stock lifted and treated at the Forest Service's nursery at Placerville, California (Roy 1966). The four treatments were (1) tops sprayed with Foli-gard, a water-soluble polymer which forms a continuous film upon drying; (2) roots dipped in Rutex 59, a solution of a placticized acrylic-type polymer which has high water-absorbing and -retaining qualities; (3) tops sprayed with Foli-gard and roots dipped in Rutex 59; and (4) control. Control seedlings had the best early-season survival, although by late August their survival averaged the same as those treated with Foli-gard (85 pct). Survival of seedlings treated with Rutex 59, either alone or in combination with Foli-gard, was about 20 percent poorer by late August. Also, Rutex 59 depressed seedling height growth, but Foli-gard significantly enhanced it.

Adverse Sites

Factors that strongly influence regeneration success include aspect, slope steepness, and soil characteristics. Heat and drought on southerly aspects often cause regeneration problems. The least favorable aspect for regeneration of plantations on the Six Rivers National Forest in northwestern California was about 12° west of south (Strothmann 1979).

The value of shade in establishing Douglas-fir regeneration from seed has been pointed out already. On hot, south-facing slopes, shade may also benefit planted Douglas-fir if stock condition is less than optimum, or if seedlings are not planted at the most favorable time.

On several test sites in California, shade decidedly benefited survival of both 1-0 and 2-0 Douglas-fir (Adams and others 1966). Shade was provided by shingles 5 to 7 inches (13 to 18 cm) wide inserted into the ground on the south-southwest side of each seedling. Shingles protruded 8 to 10 inches (20 to 25 cm) above ground and were slanted so that tops were directly over the seedlings. The 1-0 stock was benefited most by shading. No unshaded trees of this age class were alive after 2 years at one locality. For seedlings of both age classes, the benefits of shading were greatest on the driest of the test sites.

Shade helps reduce severe moisture stress, which seedlings may otherwise sustain on south-facing clearcut blocks (Lindquist 1977). Douglas-fir seedling moisture stress, survival, and growth in response to aspect and overstory canopy were studied on granitic soils on the Six Rivers National Forest in Northern California. Seedlings on clearcut blocks had plant moisture stress (PMS) values that exceeded 15 atmospheres at midmorning in June, but seedlings on partial-cut plots did not exceed this level until September. PMS values were highest on the clearcut plot with a south aspect, reaching predawn stresses of 21 atmospheres by September. This plot was the only one on which stocking was inadequate, suggesting that leaving a partial overstory on hot, dry slopes may help survival. Height

growth, on the other hand, was best on the south-facing clearcut, indicating that established seedlings grow best in full sunlight.

Type of shade has been found to influence seedling survival. In southwestern Oregon (Minore 1971), planted Douglas-fir seedlings were observed growing under three conditions: (1) in the open with no shade; (2) in the open, but shaded artificially with rocks, logs, or bark (dead shade); and (3) under existing brush (live shade). After two growing seasons, only 10 percent of the unshaded seedlings were alive compared with 47 percent under brush and 60 percent under dead shade. Although dead shade produced the best survival, providing it was time-consuming and expensive. Thus, under some circumstances, planting under existing brush may be the best compromise if the brush species does not compete too severely for soil moisture and nutrients.

The shady side of large cull logs often appears to be a good planting spot; but if logs still have bark on them, they are a potential hazard (Roy 1955b). About 3 years after logging, the bark of cull Douglas-fir logs begins to slough off in large patches. Seedlings upon which these patches fall generally are killed by smothering.

Even though some studies demonstrate the benefits of shade, shade is not essential for successful regeneration of south-facing clearcut blocks when all other facets of a planting operation are favorable. On the Hoopa Indian Reservation in northwestern California, shade of several intensities did not significantly improve survival of either 1-0 or 2-0 Douglas-fir seedlings planted on a 50 percent south-facing clearcut slope (Strothmann 1972). Survival after 2 years was good (83 pct or better) for all treatments, from 0 to 75 percent shade. Good survival was attributed to good planting stock, careful planting, a deep loamy soil, and the removal of competing vegetation. Although shade did not significantly affect survival, it reduced the growth of the planted trees. Best growth (in terms of height, stem diameter, weight of top, weight of roots, and root length) generally was associated with the least shade.

In the Siskiyou Mountains of southwestern Oregon, Douglas-fir grew best in full sunlight, and rarely grew where light intensity was less than 10 percent (Emmingham and Waring 1973). Maximum shoot elongation generally was found on trees in bare areas with little competing vegetation, or where seepage water was available.

A different approach for improving survival of Douglas-fir on hot, dry sites is selecting strains with inherently high drought resistance. An Oregon study found a decided difference in this characteristic between Douglas-fir seedlings from a moist coastal site (mean annual precipitation of 60 inches [152 cm]), and a dry inland site (mean annual precipitation of 20 inches [51 cm]) (Heiner and Lavender 1972). Seeds from the moist area were collected from numerous trees (though from a single stand) and pooled as a single lot. Five seed lots from the dry area were collected from five individual trees. Seeds were sown in a nursery near Corvallis, and 2 years later some seedlings from each lot were lifted and transplanted to a lysimeter where moisture conditions were controlled. The soil was wetted to field capacity at time of transplanting, and then allowed to dry naturally during the growing season. Rain was excluded by a plastic cover.

At the end of the growing season, seedlings from the dry source showed significantly better survival (55 to 70 pct) than those from the humid source (16 pct). Also, seedlings from the dry source all had much earlier and more complete bud burst than did those from the humid source. Differences in shoot:root ratio or cuticle thickness among seedlings of the six different lots were not significant.

A word of caution should be added, however, about taking seedlings that are genetically adapted to one environment and planting them in another. In most instances, the fundamental rule that regeneration should be accomplished with planting stock from local seed sources is still the safest guide and normally will produce the best long-term results.

Modifications of the planting technique comprise another approach for improving the survival or growth of trees planted on hot, droughty sites. One modification is deep planting—planting the seedling deeper in the ground than it grew in the nursery. The primary purpose is to get the roots deeper into the soil, where more moisture is usually available. A secondary purpose is to bury some of the foliage to reduce transpiration losses during the critical period of establishment. This technique has been used most often in the droughty sites of the southern pine region where it frequently has resulted in improving height growth (McGee and Hatcher 1963, Shoulders 1962, Slocum and Maki 1956).

In a trial in southwestern Oregon, 2-0 Douglas-fir was deep-planted in hopes of improving survival in an area where normal planting success was poor (U.S. Dep. Agric., Forest Serv. 1957). Survival did not improve, however, and averaged only 38 percent for both normal and deep planting.

In northern California, deep planting of 2-0 Douglas-fir at two different depths was tested against planting at normal depth, on a hot south-facing slope with a gravelly loam soil (Strothmann 1971a). In one treatment the trees were planted with 25 percent of their stems buried, in the other with 50 percent buried. Neither treatment significantly improved survival over that of normal planting, but early height growth benefited, especially when 50 percent of the stem was buried. Although the superior growth rate continued for 3 years after planting, the deep-planted trees took this long to overcome their initial height handicap and catch up to the trees that were planted at normal depth. After 10 growing seasons, differences in survival attributable to planting technique were not significant (Strothmann 1976). Also, height growth differences among the several planting methods were no longer significant. Thus, deep-planting Douglas-fir in southwestern Oregon or northwestern California produced no long-term benefit.

Another modification of bare-root planting is "sandwich" planting, a technique tried with 2-0 Douglas-fir seed-

lings on the Lower Trinity Ranger District of the Six Rivers National Forest (Schubert and Roy 1959). This method consisted of encasing a seedling's roots between two pieces of stiff, water absorbent, fibrous material (about 3 by 8 inches [7.6 by 20.3 cm]) stapled together. And the stiffness of the sandwiches forced workers to set seedlings in the ground properly, because the roots could not be bent in shallow planting holes. The control was bare-root seedlings planted with planting hoes. Similar stock was planted in sandwiches that had been soaked in water for 4 hours, and seedling roots had been puddled in a slurry of forest soil (Hugo clayey loam). The third treatment was the same as the second, except that the sandwich material was soaked in a fertilizer solution ($1^1/_2$ oz [42.5 g] of 7-9-5 fertilizer per gallon [3.78 liters] of water) and seedling roots were puddled in a fertilized slurry of forest soil (18 oz [510 g] of uramite, 38-0-0, added to 4 gallons [15.14 liters] of mud).

Planting Douglas-fir in sandwiches without fertilizer at first seemed beneficial, but any early-season benefit vanished by July 28: the soil near the surface dried, and the roots probably did not grow out of the sandwiches and deeper into the soil where moisture was still available. Final results showed no advantages for sandwiches over bare-root planting. The greatest mortality was found among seedlings in sandwiches to which fertilizer had been added.

Plantings sometimes fail on certain soils because workers have difficulty preparing an adequate planting hole with conventional tools such as the hoedad and planting bar. Power soil augers, which are becoming increasingly popular, overcome this problem. Augers have been used operationally on some National Forests and other ownerships for more than a decade, and many of the earlier problems associated with them have been solved. The newer models are safer to use and mechanically more reliable than earlier models; and they perform better in rocky soils. Some users have reported about 15 percent better survival of trees planted with the auger than with conventional hand tools.

On steep ground, dry ravel and debris movement sometimes kill seedlings. One possible solution tested on the H. J. Andrews Experimental Forest in western Oregon is using large 3-0 planting stock (Berntsen 1958). The test site was situated on a slope of about 50 percent at 2000 feet (610 m) elevation. Significantly more 3-0 Douglas-fir stock survived than did the normally planted 2-0 seedlings at the end of both the first and second growing seasons. Movement of surface materials (soil, rock particles, litter, and rotten wood) killed many more 2-0 seedlings (23 pct) than 3-0 seedlings (8 pct) during the first season.

Other practices that minimize seedling loss caused by soil and debris movement include these (Franklin and Rothacher 1962): planting on the flattest available microsites, such as uphill from stumps and uprooted trees; and planting near the outside rather than the inside edges of benches. Other favorable planting spots are the downslope side of stumps, well-anchored rocks, or other objects which help divert moving debris from seedlings.

Containerized Stock

Container-grown seedlings have been planted for many years for landscaping and for certain special forestry applications such as erosion control. However, their widespread use in reforestation programs is more recent.

Container-grown seedlings have both advantages and disadvantages. One major advantage is the negligible disturbance to the seedling's root system when outplanted. Even under the best conditions, bare-root stock undergoes "planting check" (a temporary reduction in top growth) when outplanted. In British Columbia, undisturbed 2-0 Douglas-fir in the nursery grew the following year at an average rate of 0.8 inch (2 cm) per inch (2.54 cm) of height at the start of the season. By contrast, the growth rate of trees that were carefully lifted and outplanted was only about 0.16 inch (0.4 cm) per inch of height at time of planting, or only 20 percent of that of the undisturbed trees (Smith and Walters 1963).

The biological soundness of containerized planting and the excellent growth potential possible with the method were demonstrated in Oregon (Owston and Stein 1972b). In early August, $1\ ^1/_2$-year-old nursery-grown Douglas-fir seedlings were potted into four types of containers; 7-inch (18 cm)-deep peat pots, 1-quart milk cartons, 10-inch-long (25 cm) plastic mesh tubes, and 10-inch-long cardboard containers that were slightly tapered and impregnated with resin. After growing several months outdoors, the trees were outplanted in late fall and also in spring, along with 1-1 and 2-0 seedlings on a site in the Cascades.

First-year height growth of containerized trees averaged about $2^1/_2$ times that of bare-root stock. Survival of all stock was relatively good, but significantly higher for the containerized trees (95 pct) than for the bare-root seedlings (83 pct). Survival of Douglas-fir seedlings did not differ significantly among container types, but height growth was best in the plastic mesh tubes, and poorest in the milk cartons. Although this study used much larger containers than those used in production planting, and the trees were not started from seed in the containers, it demonstrated the basic advantage of moving a tree to a field environment without major disturbance of its root system.

In a British Columbia study of experimental plantations on a variety of forest sites, the biological performance of $4\ ^1/_2$-inch (11.4 cm) Walters' bullet, a $4^1/_2$-inch plug, and bare-root stock were compared (Arnott 1971). After 3 years, on medium elevation sites, the average survival (percent) of Douglas-fir seedlings was plug, 84; bare-root, 80; and bullet, 68. After 2 years, on high-elevation sites, survival of bare-root seedlings was best (97 pct), with survival of plug seedlings also good (85 pct).

One of the claimed benefits of using containerized stock is it lengthens the normal planting season, allowing both earlier and later planting than would be possible with bare-root stock. Several trials in Humboldt County, California, evaluated the effect on tree survival of planting as early as September or as late as May. The recommended planting season for bare-root stock in this area is mid-December to

mid-March (Schubert and Adams 1971). Containerized Douglas-fir seedlings were planted on two coastal sites and two sites a few miles inland during two planting seasons (1974-75 and 1975-76). Precipitation was nearly normal the first season, and about 15 percent below normal the second season.

First-year survival exceeded 83 percent for the September plantings on both coastal sites, but was only about 75 percent on one interior site and less than 60 percent on the other. October plantings had better survival, averaging about 87 percent for the inland sites and 93 percent for the coastal sites. The May plantings in the season with normal rainfall had excellent first-year survival (91 pct) for both the inland and the coastal site. May planting in the drier year, however, resulted in lower first-year survival on both the coastal site (79 pct) and the inland site (60 pct). By comparison, first year survival of trees planted in January exceeded 92 percent for all four sites and for both planting seasons. Whether bare-root seedlings might have performed similarly is not known, because they were not included in this study.

The findings suggest that the use of containerized Douglas-fir will permit moderate extensions of the normal planting season without significantly increasing mortality on coastal sites. However, on inland sites with drier climatic regimes, or during years of subnormal rainfall, such extensions may reduce survival by as much as 30 percent.

Many questions concerning the relative merits of container-grown versus bare-root seedlings remain unanswered. Some answers should be forthcoming within the next few years when the results of various field trials currently underway are analyzed and published. Some earlier field trials may have compared the two types of stock unfairly because containerized seedlings were relatively small and bare-root seedlings were much larger (Stein 1976). In these comparisons on severe sites in the Pacific Northwest, container stock usually has not matched the performance of bare-root stock. Tissue differences between young containerized seedlings and older, more woody, bare-root seedlings may put the former at a disadvantage with respect to animal damage. In several instances, container-grown seedlings have been damaged more heavily than adjacent bare-root seedlings (Stein 1976). Our present state of knowledge indicates that either of these two planting methods will work if their basic requirements are not compromised, and if each is used under conditions for which it is best suited.

CARE OF YOUNG STANDS

Planting trees or getting seedlings to germinate is far from the final step in regenerating an area. Starting almost immediately, numerous problems beset the young trees. Some of these problems—such as desiccating winds, prolonged drought, untimely frosts, or other meteorological variables—are, for all practical purposes, beyond the forester's control. Other factors are amenable to varying degrees of control, though effective control may be limited at times by environmental constraints, lack of adequate technical information, or simply by costs that are too high to be economically feasible.

Basically, the factors over which the forester has some measure of control are those which involve plants or animals. Because of intertwining ecological relationships, one factor can sometimes be controlled by controlling or manipulating the other.

Reducing Vegetative Competition

Whether vegetative competition will pose a serious threat to conifer regeneration in a given plantation depends on many factors, including the plant species present, their density and abundance, the fertility of the site, how wet or dry it is, and how thoroughly the site was prepared. Competition from grass will be discussed first, not because it is necessarily of greatest importance, but because treatment is required soon after planting where potential problems exist.

In Oregon, where the competing vegetation was grass, first-year survival of 2-0 Douglas-fir was considerably improved by application of atrazine (Bickford and Hermann 1967). It was applied at three rates ($1^2/_3$, $3^1/_3$, and 5 lbs per acre [1.87, 3.73, and 5.60 kg/ha]) at the end of March, after planting on five different dates during the preceding fall and winter. A fourth series of plots received no atrazine. At the end of the first growing season, the lowest application rate had increased survival twofold, and the two higher rates had increased it nearly fivefold over that on the untreated plots.

In southwestern Oregon, 10 chemicals were tested to determine their relative effectiveness for grass and forb control in three plantations of young Douglas-fir 6 to 15 inches (15 to 38 cm) tall (Gratkowski 1976). Two study areas were on wet sites on the Siskiyou National Forest, and the third was on a dry site in the foothills of the Cascade Range east of Roseburg. Atrazine at 4 pounds a.i. (active ingredient) per acre (4.5 kg a.i./ha), and Terbacil at 2 pounds a.i. per acre (2.2 kg a.i./ha) were the most effective. Foliage sprays of both chemicals controlled grasses without permanent damage to Douglas-fir seedlings. Terbacil damaged a few conifers during the first summer after spraying, but no trees were killed and all were healthy and vigorous at the end of the second summer. Granular Dichlobenil at 4 to 6 pounds a.i. per acre (4.5 to 6.7 kg a.i./ha) was excellent in controlling grasses and forbs for two summers, but it consistently damaged young Douglas-fir trees.

Sometimes, several herbicides in combination are more effective in controlling grass competition than are the same

herbicides applied singly. The appropriate combinations also may be less injurious to Douglas-fir seedlings. Dalapon injured Douglas-fir when used alone, but was applied safely with beneficial effects when combined with atrazine and 2,4-D at rates of up to 3 (atrazine) and 4 pounds (2,4-D) per acre (3.36 and 4.48 kg/ha) (Newton and Overton 1973). In these combinations, dalapon did not harm Douglas-fir seedlings, even when applied at rates up to three times that needed for adequate control of grasses. Combinations of dalapon, atrazine, and 2,4-D also have successfully controlled weeds in two seed orchards, and in a reforestation study in Mendocino County, California (Adams 1979).

Although grass and forbs can be serious competitors—especially when conifer seedlings are small—brush is generally a more serious long-term problem, partly because many brush species reproduce by sprouting when the older stems are cut or damaged. The new sprouts often can outgrow associated conifer seedlings for 10 to 15 years or longer.

The height growth of sprouts of five of the more common hardwood brush species that often occupy forest land in northwestern California after fire or logging was measured (Roy 1955a) *(table 1)*. In descending order of height growth, the species were bigleaf maple, Pacific madrone, Oregon white oak, tanoak, and Pacific dogwood. Within each species, the greater the diameter of the parent tree, the more numerous, taller, and the larger were the sprouts. Bigleaf maple is restricted mainly to streambanks or other places with abundant soil moisture. Dogwood grew in a separate area from the others, and was the only species subjected to heavy browsing by deer (Roy 1955a). Although the growth rate of tanoak sprouts was exceeded by three of these species, it is the most important competitor because of its longevity, ultimate size, and widespread distribution throughout the region.

Effect of Brush Competition

Opinions differ among foresters as to the extent to which brush may retard the growth of conifers. That the answer depends on the species involved seems clear. Few studies in the Klamath Mountains Region have specifically tested the degree to which various species of brush retard growth of young Douglas-fir.

Table 1—*Average growth characteristics of sprouts in five hardwoods that commonly compete with Douglas-fir, after three growing seasons, northwestern California*

Species	Sprouts		
	Height	Crown diameter	Number in clump
	Feet (meters)		
Bigleaf maple	13 (4)	15 (4.6)	37
Pacific madrone	10 (3)	8 (2.4)	13
Oregon white oak	9 (2.7)	8 (2.4)	10
Tanoak	7 (2.1)	7 (2.1)	12
Pacific dogwood	6 (1.8)	5 (1.5)	14

In one study on Brush Mountain on the Lower Trinity District of the Six Rivers National Forest, the site was tractor-logged in 1952 and prepared that same fall by broadcast burning. Douglas-fir 2-0 seedlings were planted the following year. In 1958, the trees on part of the area were released by hand-cutting competing shrubs—primarily tanoak, madrone, and snowbrush. The resprouting shrubs on the released portion were sprayed in fall 1959 with 2,4,5-T.

In the 17 years after the initial release in 1958, the released trees steadily increased their height advantage over those not released. At the outset they had the same average height—4.5 feet (1.37 m). For the first 4 years the gain from release was small, but it increased with time:

Years after release:	Height gain by released trees
	Feet (meters)
4	0.5 (0.15)
6	2.9 (0.88)
8	5.0 (1.52)
10	6.9 (2.10)
17	12.4 (3.78)

After 17 years, the released trees averaged 38.4 feet (11.70 m) tall and the unreleased only 26.0 feet (7.92 m). The two groups also differed in tree vigor. Criteria of vigor were needle complement, needle color, and needle length. When the study began, all trees were classed as vigorous. After 17 years, all released trees were still in this category, but only 55.6 percent of the unreleased trees were classed as vigorous, 18.5 percent as medium in vigor, and 25.9 percent as poor.

As to the relative severity of competition afforded by snowbrush and tanoak, Douglas-fir trees began to grow through the snowbrush in 1964—12 years after logging and the same year that some snowbrush plants died. Individual snowbrush plants have continued to die since that time, especially following severe damage from snow breakage during winter 1964-1965. Tanoak, on the other hand, is providing increasingly severe competition.

A 5-year study in southwestern Oregon indicated that a dense overstory of varnishleaf ceanothus definitely retards height growth of young Douglas-fir in the understory (Gratkowski 1969). All degrees of release from varnishleaf ceanothus resulted in increased height growth of the conifers, and complete exposure to sunlight provided the longest lasting release for young Douglas-fir. On plots where varnishleaf ceanothus had been basal-sprayed, growth of Douglas-fir was 1.3 to 1.5 times that of similar trees under unsprayed live brush. A similar increase in growth of Douglas-fir was noted under standing dead ceanothus on aerially sprayed areas. Growth of completely released trees, where ceanothus was lopped and stump-sprayed, was more than 1.8 times that of trees under live ceanothus.

The beneficial and detrimental influence of snowbrush ceanothus on several species of conifers has been studied on the west slope of the Oregon Cascades (Zavitkovski and others 1969). The survival and height growth of both

planted and natural Douglas-fir seedlings were observed in 12 stands of snowbrush, which ranged in age from 0 to 15 years. The two benefits sometimes claimed for snowbrush—its nitrogen-fixing capacity and its functioning as a nurse crop—were more than offset by its suppressing effect on seedling growth. Only when planted immediately after slash burning as 2-0 or 2-1 stock of local origin could Douglas-fir overcome the competitive challenge of the developing snowbrush.

Snowbrush and red alder in Oregon have different effects on Douglas-fir (Newton 1964). On a typical xeric site in the Cascade Mountains, open-grown Douglas-fir grows more than twice as fast as Douglas-fir under a canopy of snowbrush. However, snowbrush reaches its maximum height in about 15 years, and, after another 3 years, suppressed, slow-growing Douglas-fir trees eventually emerge through the brush canopy. In contrast, red alder grows so rapidly on a mesic coastal site that it hopelessly outdistances Douglas-fir competing with it. In only 7 years, alder may attain a height of 40 feet (12 m), whereas the height of Douglas fir, even without competition, would be less than 10 feet (3 m) at this age. The possibility of Douglas-fir overcoming such an unfavorable initial position for light is remote, and the only trees with any chance of surviving without assistance are those in openings.

The impact of vegetative competition on planted Douglas-fir also was studied on a tract near the Oregon coast, about 7 miles (11.3 km) southeast of the town of Florence (Ruth 1956). The area was clearcut in 1947, and slash was broadcast-burned that fall. The burn was excellent, but because of poor Douglas-fir seed crops in both 1947 and 1948 the anticipated natural regeneration failed to materialize. By the time the area was planted in March 1949 with 3-0 Douglas-fir seedlings, brush resprouting was well underway. Three months after the Douglas-fir seedlings were planted, competing vegetation averaged 2 feet (0.6 m) tall. Five years later it averaged 5 to 6 feet (1.5 to 1.8 m) tall compared with an average height of slightly over 3 feet (0.9 m) for the Douglas-fir. After the fifth growing season, the principal competing brush and tree species listed in declining order of importance were salmonberry, western thimbleberry, red alder, Pacific red elder, willow, and vine maple.

Increasing shade intensity to 75 percent did not reduce Douglas-fir survival, but progressively reduced height growth. When shade intensity exceeded 75 percent, both height growth and survival were adversely affected. At the end of five growing seasons, annual height growth of overtopped trees averaged only 8 inches (20 cm) compared with 25 inches (63 cm) for unshaded trees.

Controlling Brush and Hardwoods

Since the late 1950's, much research has been directed toward developing techniques for effectively controlling brush and hardwood competitors of young Douglas-fir. Most of this work has dealt with chemical control methods because of their biological effectiveness and their cost advantages over other methods. In recent years, however, increasingly stringent regulations have eliminated some of these chemicals from the forester's choices, and have restricted considerably the use of others. Consequently, a growing interest has developed in alternative methods of conifer release—especially manual methods, such as hand-cutting of brush.

Tanoak and Pacific Madrone—Douglas-fir can be released effectively from tanoak and madrone by an aerial application of a low volatile ester of 2,4-D in early spring just before or during budbreak of Douglas-fir (Gratkowski 1961a, 1975). The recommended dosage is 3 pounds a.e. (acid equivalent) per acre (3.36 kg a.e./ha). If necessary, a second, milder application of 1.5 to 2 pounds a.e. per acre (1.68 to 2.24 kg a.e./ha) can be applied 1 or 2 years later. Research in California showed that late fall application of low-volatile esters of 2,4-D and 2,4,5-T effectively controlled tanoak sprouts. Sprout clumps sprayed after mid-October did not resprout, whereas some clumps sprayed in July and August did (Roy 1956).

Tanoak and madrone also can be controlled effectively by treating individual stems with the cut-surface (hack-and-squirt) method. In northern California, stems of tanoak and madrone 3 to 4 inches (7.6 to 10.2 cm) in diameter were treated by severing the bark around the base of each tree with a hatchet (Radosevich and others 1976). Each cut then received approximately 0.06 ounces (2 ml) of either undiluted 2,4-D, 2,4,5-T, or picloram. Some trees were left untreated. Ten years later, mortality of the treated hardwoods in response to application of these chemicals was as follows: 2,4-D, 86.6 percent; 2,4,5-T, 78.6 percent; and picloram, 94.1 percent. None of the untreated hardwoods had died. Basal area growth of the released understory Douglas-fir trees, expressed as a percent of growth of the unreleased trees, was increased by 260 percent (2,4-D), 451 percent (2,4,5-T), and 405 percent (picloram). The relatively lower growth response of the conifers in the 2,4-D treatment was attributed to greater mutual competition among the Douglas-fir trees.

Triclopyr has proven to be an effective herbicide for control of tanoak, both as a cut-surface treatment on standing trees and for application to freshly cut stumps to reduce sprouting (Warren 1980). Two growing seasons after treatment with triclopyr applied to cuts in the cambium on centers 4 to 5 inches (10 to 13 cm) apart, 94 percent or more of the treated tanoak trees were killed. None of the intermingled Douglas-fir trees showed any evidence of herbicide damage. Scrub tanoak is classified as "resistant" to chemical treatment (Gratkowski 1959). Of six herbicides tested, the most effective were low-volatile esters of 2,4-D and 2,4,5-T, but even these caused only partial die-back of the stems and branches. No treated plants were killed.

Ceanothus—*Ceanothus* species commonly regarded as competitors of young Douglas-fir are snowbrush, varnishleaf ceanothus, mountain whitethorn, blueblossom, and deerbrush. Of these, only deerbrush is rated as highly susceptible to chemical control with a foliage spray of low-

volatile esters of 2,4-D in water (Gratkowski 1959, 1978). The other species are less susceptible to chemicals, but nevertheless are affected significantly by low-volatile esters of 2,4,5-T sprayed on foliage from early spring throughout the growing season (Gratkowski 1975). This treatment kills aerial parts, but does not eliminate resprouting, although it greatly curtails it for varnishleaf ceanothus (Gratkowski 1978).

Chinkapin and Canyon Live Oak—Two forms of chinkapin—giant (or golden) chinkapin and golden evergreen-chinkapin—are recognized as competitors of Douglas-fir in portions of the Klamath Mountains Region (Gratkowski 1959, 1978). Both forms are regarded as resistant to chemical control—i.e., only parts of the stems and branches die back after being sprayed with herbicides. Low-volatile esters of 2,4,5-T are the most effective for releasing Douglas-fir from these species.

Canyon live oak is another common competitor that also is resistant to chemical control. Unlike the chinkapins, however, it was more susceptible to low-volatile esters of 2,4-D than to 2,4,5-T, but neither chemical achieved complete top kill (Gratkowski 1959, 1978).

Manzanita—Among species of manzanita that compete to varying degrees with Douglas-fir seedlings, three are readily killed with low volatile esters of 2,4-D (Gratkowski 1959). These include hairy manzanita, hoary manzanita, and Howell manzanita—sometimes identified as the non-sprouting form of greenleaf manzanita (Gratkowski 1978). The sprouting form is by far the most important species of manzanita in the Coast Ranges and Siskiyou Mountains, not only because of its wide range and abundance, but also because of its resistance to silvicultural control (Gratkowski 1978). Greenleaf manzanita forms a conspicuous burl at the root crown, and when the aerial parts are killed by cutting, fire, or chemicals, numerous new sprouts develop from dormant buds in the burl. The recommended treatment for controlling this species is a foliage spray of 2,4-D in a light oil-in-water emulsion (Gratkowski 1978).

Salmonberry and Western Thimbleberry—Salmonberry and western thimbleberry, found on coastal Douglas-fir sites in California and Oregon, are classed as being only moderately susceptible to herbicides (Gratkowski 1978). The recommended control for areas where both species grow is a late foliar spray of low-volatile esters of 2,4,5-T applied in an emulsion carrier (Gratkowski 1971, 1975). On salmonberry alone, amitrol-T was more effective than 2,4,5-T in both early and late foliar sprays (Stewart 1974a). However, amitrol-T does not control western thimbleberry. Because most stands of salmonberry also contain thimbleberry, the use of amitrol-T frequently releases thimbleberry, thus permitting it to increase its occupancy of the site.

Red Alder and Vine Maple—The treatment for effectively releasing Douglas-fir from red alder depends upon the height of the alder when treated (Gratkowski 1975). If the alder is 15 feet (4.6 m) or less in height, a single budbreak spray usually will be sufficient. If the alder exceeds 15 feet in height, two sprays may be needed: a budbreak spray and a late foliar spray. For the budbreak spray, a mixture of 2,4-D and 2,4,5-T in diesel oil is recommended; and for the late foliar spray, only 2,4-D in water (Gratkowski, 1975). The foliar spray should be applied 1 to 2 years after the initial spray, or when the alder has enough foliage to intercept most of the spray.

Acceptable kill to release Douglas-fir from vine maple can be obtained with early spring aerial application of low-volatile esters of 2,4,5-T in diesel oil (Gratkowski 1975, Stewart 1974b). The spray should be applied just before or during budbreak of Douglas-fir. As with alder, an oil carrier is necessary for budbreak spraying because these species are then leafless, and the herbicide must enter through the bark.

Douglas-fir Sensitivity to Herbicides

The sensitivity of conifers to damage by herbicides must always be considered. In southwestern Oregon, 2,4-D and 2,4,5-T had no pronounced effect on Douglas-fir (Gratkowski 1961b). However, season of application and the carrier used had major effects on damage and mortality. With all tested formulations, the amount of damage from autumn application was noticeably less than from the same spray applied in summer. Also, adding oil to the carrier caused a substantial increase in damage and mortality when the chemicals were applied during the summer. In autumn, however, Douglas-fir was relatively resistant to either herbicide—even in emulsion carriers.

Similar results were obtained in a study that tested the seasonal tolerance of six coniferous species to eight foliage-active herbicides (Radosevich and others 1980). Each herbicide was applied on three different dates: April 14, July 7, and September 23, 1977. The trees were still dormant at the first application, were actively growing during the second application, and had ceased seasonal growth before the September application. All conifer species were most tolerant when herbicides were applied in September, and most susceptible to damage when the chemicals were applied in July. No injury to Douglas-fir was observed when either 2,4,5-T or diclorprop was applied in the fall, although some mortality resulted from fall application of Silvex. Also, fall application of glyphosate caused some injury to Douglas-fir.

The seasonal nature of herbicide selectivity in coniferous species probably is caused by differential absorption through leaves and restricted distribution after absorption (Radosevich and others 1980). As conifer leaves become more sclerophyllous with age, less herbicide is likely to penetrate them—an effect enhanced as the growing season progresses, because herbicide distribution would be restricted as moisture stress increases.

Preventing Animal Damage

Deer

Among the animals that damage or kill young Douglas-fir trees in northwestern California, deer rank as the most

injurious agent (Packham 1970). Mule deer and black-tailed deer are both subspecies of the only species *(Odocoileus hemionus)* native to California. In the Pacific Northwest, deer are responsible for most animal damage to reforestation (Black and others 1969). Such damage by deer represented about 56 percent of all animal damage recorded.

The apparent interrelationship among brush, deer, and Douglas-fir was investigated in three areas in northern California: Slate Creek, about 7 miles (11.3 km) north and slightly east of Weaverville; Brush Mountain, about 3 miles (4.8 km) west of Salyer; and the Swanson Unit, about 2 miles (3.2 km) east and 1 mile (1.6 km) south of Salyer (Roy 1960b). At Slate Creek, deer browsed lightly on conifers because of the abundance of bigleaf maple seedlings and dogwood sprouts, which they prefer.

At Brush Mountain, deer browsing on Douglas-fir seedlings was moderately severe because more highly preferred browse species were not abundant. Repeated browsing reduced height growth appreciably. Trees browsed four times during the 6 years after planting lost 2.3 years of growth; those browsed five times lost 2.7 years of growth.

On the Swanson Unit, proximity to tanoak sprout clumps afforded Douglas-fir seedlings some physical protection from browsing. Six years after planting, deer browsing had reduced the height of unprotected seedlings by an amount equivalent to 3.6 years of growth compared with seedlings protected by brush clumps. The protected seedlings averaged 39.7 inches (101 cm) in height; unprotected seedlings averaged only 11.0 inches (28 cm).

On the Coast Range of Oregon, browsing by black-tailed deer, and also by mountain beaver and perhaps brush rabbits, reduced height growth of Douglas-fir (Ruth 1956). Some trees were not browsed at all, some only once, and others two or three times. The average heights of the surviving trees after 5 years were as follows: not browsed, 55.9 inches (142 cm); browsed once, 40.1 inches (102 cm); browsed twice, 16.5 inches (42 cm); and browsed three times, 8.3 inches (21 cm). Few trees were killed by browsing alone; but reduced height growth and loss of vigor from browsing contributed to overtopping by brush and herbs, and many trees subsequently died.

Substantial retardation of height growth of Douglas-fir as a result of deer browsing also was noted in western Washington (Hartwell 1973b). On each of three plantations that had already been subjected to deer browsing for 1 to 5 years, some trees were caged to prevent further deer browsing, and others were left uncaged. At the beginning of the study, the trees on each plot were uniformly shrub-like and slightly over 1 foot (30 cm) in mean height. After 6 years, the mean height of the caged trees was about $2^1/_2$ feet (0.76 m) greater than that of the uncaged trees on each of the study areas.

Although some evidence suggests that browsing may reduce height growth of seedlings significantly as long as the terminal shoots remain within reach of the deer, other evidence suggests that after terminals grow out of reach, browsed trees tend to catch up to their unbrowsed counterparts. This possibility was explored during a 16-year period on the Brush Mountain Unit near Salyer on the Six Rivers National Forest in northwestern California. Browsing began during the second growing season and peaked in the fourth season when 59.4 percent of the trees were browsed. The maximum height of browsing was 5.83 feet (1.78 m). Almost 18 percent of the trees were never browsed, but over 57 percent were browsed one to three times, and 25 percent were browsed four or more times. Two trees were browsed in 10 of the 16 years.

Sixteen years after trees were planted, browsing up to six or seven times appeared to have no effect upon their height. Browsed trees, which had fallen behind in height growth at 6 years after planting had, in general, caught up to unbrowsed trees. In fact, browsing up to five times seemed to stimulate height growth. However, until it is known whether browsed trees have some nutritional advantages over unbrowsed trees, a cause-and-effect relationship should not be assumed.

Even heavy browsing is unlikely to affect seriously either tree volume or quality of Douglas-fir at rotation age (Mitchell 1964). Average reduction in tree height attributable to animal feeding in heavily browsed plantations varied from $^1/_2$ foot to 2 feet (0.15 to 0.61 m) over an 8- to 10-year period, on Vancouver Island, British Columbia. Exposed trees were browsed more heavily than those protected by vegetation or logging slash.

An Oregon study led to similar conclusions: deer browsing will probably be of little consequence in reducing growth over the length of an entire rotation (Crouch and Paulson 1968). Over a period of 7 years, however, the trees that had been protected from browsing since time of planting averaged about $1^1/_2$ feet (0.46 m) taller than the unprotected trees. The unprotected trees showed no sign of catching up—even though they were not browsed during the last 3 years of observation.

Until the effects of deer browsing on quantity and quality of Douglas-fir yield are determined more clearly, land managers have several choices for coping with the problem of browsing damage. Such measures include the use of mechanical barriers, chemical repellents, habitat modification through cultural practices, and genetic modification.

Mechanical Barriers—Mechanical barriers have proven effective. They can be high fences that exclude deer from an entire plantation, small cages that protect individual trees, or even smaller devices that protect only the terminal leader or terminal bud during a critical stage. The chief disadvantage of the larger mechanical barriers is cost, because they must be installed, maintained, and—in most cases—eventually removed (Longhurst and others 1962). Therefore, most interest has been directed toward the smaller mechanical devices, such as protective sleeves for terminal shoots.

Plastic sleeves have performed well in protecting Douglas-fir seedlings from browsing damage (Campbell and Evans 1975). They can be formed from various materials and tailored to meet specific needs. Polyethylene sleeves are available in either rigid or flexible form, in different colors, and biodegradable at different rates. One type degrades

within 1 year; other types remain intact for periods up to 10 years or more.

Mechanical and chemical means of deterring browsing of Douglas-fir were compared on the Tillamook Burn in western Oregon (Hines 1971). The effectiveness of a polyethylene sleeve slipped over the terminal shoot, and that of tetramethyl thiuram disulfide (TMTD), a commercial repellent, were tested inside a deer enclosure where heavy browsing pressure could be maintained. In both years of observation, terminal browsing of untreated trees was significantly greater than that of trees protected by either of the two treatments. The polyethylene sleeves provided significantly better protection than the repellent.

Chemical Repellents—One of the chemicals tested for the ability to discourage deer from browsing on young conifers is BGR (Big Game Repellent)—a formulation of putrefied eggs. It has met with varying success, with degree and duration of protection influenced by weather and other factors. Most externally applied repellents such as BGR and TMTD, however, share a common drawback—they afford relatively little protection to newly elongating shoots.

A systemic repellent that is readily absorbed and translocated to all parts of a seedling, including the newly developing shoots, is needed. To date no completely successful systemic has been found. One that showed promise is OMPA (octamethyl-pyrophosphoramide) (Rediske and Lawrence 1964). First symptoms of phytotoxicity were observed at 1500 ppm, with lethality reached between 3000 and 4000 ppm tissue. OMPA was translocated readily in seedlings, both up and down the stem, when applied to the roots or the foliage. Bioassays indicated that the material was effective as a systemic repellent at foliage concentrations of 400 to 1000 ppm. Test rabbits clearly differentiated between untreated and treated seedlings, which they rejected as food. Thus, because the threshold for effective repellency is approximately 400 ppm, and because it is not phytotoxic to the seedling at concentrations below about 1500 ppm, OMPA showed promise of being an effective systemic repellent. The test did not include deer, however, and when others tested OMPA, the results were often disappointing.

Another way to discourage deer from browsing Douglas-fir trees is to spray a spatial repellent. One of the more promising materials of this type is putrefied fish. Among five materials tested, putrefied salmon gave the best results (Hartwell 1973a). Applying the material to balls of cotton placed about 10 inches (25 cm) below terminal buds when bud swelling began in late April greatly reduced deer browsing of the rapidly growing terminal shoot after bud burst. However, the spatial aversion resulting from this technique did not extend to nearby untreated trees, nor did it last to the end of the growing season.

The possibility of reducing animal damage by manipulating the habitat may have increasing appeal. By appropriate cultural treatments, the habitat can be made less desirable for a given animal species and more suitable for a given plant species. In a study aimed at achieving these ends, combinations of several herbicides (atrazine, dalapon, Silvex, and 2,4-D) were applied to one-half of each of three clearcuttings 30 to 70 acres (12 to 28 ha) in size, and from 8 to 12 years since logging (Borreco and others 1972). Two of the study areas were located south of Cottage Grove, Oregon, and the third was a few miles southeast of Kings Valley, Oregon. All were characterized by uniform, gentle topography, southerly exposures, and heavy deer usage.

The application of herbicides caused profound changes in vegetation. Grasses were reduced, but growth of shrubs and trees was promoted. Survival and growth of Douglas-fir were greater on the treated plots. Changes in vegetation affected the seasonal pattern of deer usage as determined by pellet-group counts, which indicated more activity on treated plots during the growing seasons. However, the amount of browsing did not differ significantly as a result of the habitat changes. Thus, both deer and Douglas-fir seemed to benefit by the altered habitat.

The possibility of luring deer away from Douglas-fir by seeding or planting other highly preferred browse species also has been tried, but results generally have been disappointing. Browse preference studies in northwestern Oregon suggest that, while the intensity of browsing of Douglas-fir seedlings may vary from season to season and year to year, this species ranks high enough on the food preference list so that some browsing can be expected regardless of the apparent abundance of other preferred forage (Crouch 1966).

Cultural Practices—Another approach to reducing the impact of animal damage on Douglas-fir involved testing whether large planting stock might be more resistant to wildlife damage than small stock because of its greater height and stem diameter (Hartwell 1973c). The tests were conducted at four localities in Grays Harbor County, Washington. The two main animal species causing damage were black-tailed deer and mountain beaver. All trees were treated with TMTD animal repellent before they were lifted from the nursery. Large stock was planted with a shovel, and small stock with a planting hoe. Wildlife feeding injuries and tree heights were recorded periodically for about 5 years after plot installation.

The study results could not be considered conclusive because only a few trees were observed. Nevertheless, the data strongly suggested that, under certain conditions, use of large stock can substantially reduce the impact of wildlife damage. When planted, the large stock varied in mean height among plots from 3.1 to 4.2 feet (0.94 to 1.28 m); the small stock varied from 1.1 to 1.3 feet (0.34 to 0.40 m). Five years later, the large stock varied from 5.5 to 11.0 feet (1.68 to 3.35 m); the small stock varied from 3.6 to 4.8 feet (1.10 to 1.46 m). The greatest benefit from using large stock was gained on plots with the densest ground cover. In areas where mountain beaver, rather than deer, posed the chief animal damage problem, even small stock had high survival with only one TMTD repellent application.

Existing evidence indicates that the application of fertilizer—specifically nitrogen—to young Douglas-fir trees increases not only their growth but also their susceptibility

to deer browsing. This theory was tested in Mendocino County, California, on a soil which was highly deficient in nitrogen, with deficiencies in phosphorus and sulfur also noted (Oh and others 1970). These elements were applied on different plots in the following forms and dosages: nitrogen as urea (45 pct nitrogen) at the rate of 100 pounds of nitrogen per acre (1.12 kg/ha); phosphorus as concentrated superphosphate (24 pct phosphorus) at 100 pounds of phosphorus per acre; and sulfur as gypsum (17 pct sulfur) at the rate of 100 pounds of sulfur per acre.

Observations showed a highly seasonal pattern of deer browsing on Douglas-fir in the study area, with heaviest browsing coinciding with the period of bud burst and rapid growth of new shoots. The preference of deer for nitrogen-fertilized trees was clear. Deer browsed 90 to 100 percent of new shoots on the nitrogen-treated plots during the first growing season, and 50 to 75 percent during the second season. On the plots not receiving nitrogen, deer browsed only 0 to 10 percent of new shoots during the first season, and 0 to 5 percent during the second growing season. Nitrogen fertilization of Douglas-fir seedlings increased the growth, protein content, acceptability to deer, and rumen microbial fermentation. Even in the second season after fertilization, the deer preferred fertilized trees, although differences in fermentability between foliage from fertilized and unfertilized trees were no longer detected.

Genetic Modification—Additional research at the Hopland Experiment Station, Hopland, California, demonstrated significant differences in rumen microbial fermentability between two Douglas-fir stocks of different genetic origin. Others have also probed the chemical makeup of seedlings from different sources in a quest for differences that may be associated with browsing preference. In western Washington, 10-year-old Douglas-fir clones resistant to deer browsing had lower dry-matter and cellulose digestibilities; essential oils with greater inhibitory action on rumen microbial activity; higher contents of fats, total phenols, flavanols, and leucoanthocyanins; and lower levels of chlorogenic acid (Radwan 1972). Results suggest that these resistance characteristics, especially the chlorogenic acid content, might be useful for screening Douglas-fir breeding stock for resistance to deer browsing.

Other Animals

Other animals can damage Douglas-fir reproduction seriously. Regionally, they damage fewer trees than do deer, but often their impact on a specific plantation can be more devastating. The pocket gopher is one of the major animal pests (Packham 1970). These animals feed on roots, stems, and branches of seedlings and saplings, usually killing the tree. They gnaw roots throughout the year, but the damage sometimes goes unnoticed until the young trees begin to lean or fade. Stems and branches are gnawed primarily in winter in areas where snow cover provides concealment. Results usually are observed the following spring or summer as completely girdled trees fade, or as the remaining stubs are found.

Several species of meadow mice can injure plantations in grassy areas by gnawing off the bark of seedlings and saplings. Girdling may extend from the root collar up the stem a foot or more, and trees several inches in diameter may be girdled. Most trees are damaged during the fall and winter in years of peak animal populations, but meadow mice are a threat to young trees at any time (Packham 1970).

Less serious animal threats to Douglas-fir regeneration, except on a local basis, are posed by the mountain beaver, elk, black bear, and the dusky-footed woodrat, although this last species could prove to be a serious threat in precommercially thinned stands. Domestic cattle and sheep also cause damage to young plantations through browsing and trampling.

GENERAL RECOMMENDATIONS

This section offers general recommendations applicable to Douglas-fir forests in northern California and southwestern Oregon. These recommendations are general guides for forest managers and other practitioners. They are based on research on Douglas-fir regeneration carried out since the early 1950's, with emphasis placed on the practical aspects of the results.

Seed Production

Apply nitrogen fertilizer to young stands in the nitrate form rather than in the ammonium form to increase seed production. Efforts to stimulate seed production in old-growth Douglas-fir by adding fertilizer have been generally unsuccessful and are not recommended. Seed production also can be increased by stem girdling. The best time to apply this technique is at the onset of flowering, but only in years when natural flowering is scant.

Site Preparation

Restrict mechanical site preparation, including use of a tractor with a rake or brush blade, to sites that are relatively flat. Tractors working on slopes steeper than about 30 percent can increase erosion on some sites. Also, on certain soil types when soil moisture is high, tractors can seriously compact the soil. On suitable sites, tractors can prepare a high proportion of plantable ground. Toothed blades can be pushed through the upper layers of soil to uproot brush while minimizing topsoil movement. Hold down costs by

keeping the tractor moving forward as much as possible, working around obstructions instead of attempting complete brush eradication.

On slopes too steep for tractors, burning now is the only feasible way to reduce the large accumulation of logging debris left on many cutblocks. Recent advances, which use helicopters to provide rapid, precise ignition, allow much better control of burning patterns and fire behavior—and greater safety for the crew—than the hand-carried driptorch method.

Severe burning (intense, prolonged heat that consumes all duff and changes the color of the upper layers of mineral soil) changes both the physical and chemical properties of soil. Whether to burn a given site must be decided carefully to determine if the improvements in ease of planting, reduction of competition, and reduction in fire hazard outweigh the possible adverse impacts on the site. On steep slopes on certain soil types, the retention of vegetation and duff is essential for preserving soil stability. Burning that destroys these protective elements may lead to increased soil movement. Also, burning stimulates germination of seeds of certain brush species, that later compete seriously with young conifers.

The chemical method of site preparation exposes no mineral soil, but effectively can retard competing vegetation. Herbicides, alone, are effective for site preparation only when the target species are highly susceptible, slash density is low, litter is light enough to permit seeding, or brush is sparse enough to allow planting at reasonable cost.

Herbicides often are used most effectively in conjunction with either mechanical clearing or prescribed burning. In dense stands of susceptible brush, herbicides can kill or weaken a substantial portion of the brush. Without additional treatment, however, the area is still unplantable because of the density of the standing dead brush. This physical impediment can be removed by mechanical clearing, burning, or both, but sufficient time should be allowed after chemical treatment to achieve maximum root kill and stem desiccation. Resprouting will be less than if either mechanical clearing or burning had been used alone.

Natural Regeneration

Natural regeneration plays only a minor part in the overall Douglas-fir regeneration effort. To keep all parts of the harvested area within reasonable distance of a seed source, limit cutblocks to less than 8 chains (160 m) wide. The side shade cast by the surrounding uncut forest also aids seedling establishment.

Various species of rodents and birds consume large quantities of Douglas-fir seed. Even if conditions are favorable for natural regeneration, high populations of seed-eating rodents or birds can prevent success unless measures, such as poisoning, are used to control these predators.

Direct Seeding

Interest in direct seeding has declined considerably in recent years because the probability of failure is relatively high and control over stocking and spacing is poor. Nevertheless, direct seeding has sufficient potential as a viable option for some landowners on some sites—if care is taken to follow the best known practices.

A mineral soil seedbed, prepared either mechanically or by a clean burn, is generally best. Douglas-fir seed can also germinate well in charcoal. Regeneration success is more likely on fine-textured than on coarse-textured soils. Sow Douglas-fir seed in late fall or early winter rather than late winter or spring. In general, northerly aspects are more favorable than southerly aspects, although good results can be produced on the latter on fine-textured soils. On southerly aspects, germination and early survival are favored by moderate shade (about 50 pct).

The same species of rodents and birds that reduce the success of natural regeneration also threaten the success of direct seeding. Therefore, do not attempt regeneration by direct seeding without providing some form of protection for the seeds. On small areas that can be hand-seeded, protect the seeds with either a repellent or poison coating, or with metal or plastic screens. The screening technique is more expensive because of the additional labor and materials required.

The only feasible method of regenerating large areas by direct seeding is to aerially broadcast repellent-coated seed. Approximately 1 pound of seed per acre is normally sufficient. Sow seeds on thoroughly prepared sites, after autumn rains have wet the upper soil horizons amply and settled loose surface soil.

Planting

Planting is by far the most common and most reliable method of regenerating Douglas-fir in the Klamath Mountains Region. Success, however, requires careful attention to detail during every stage of the process, from using the proper seed source to field planting. Seed from the proper source is essential to ensure that the planted trees will be adapted genetically to the environmental conditions at the planting site.

Lift bare-root seedlings from nursery seedbeds only when fully dormant. Dormancy in Douglas-fir is seed-source dependent. Some seed sources have a wide "lifting window" and others a very narrow one. Seedlings can be placed in cold storage until planting time. Cold storage under proper conditions does not impair seedling vigor or field survival if the seedlings are lifted when fully dormant and have a high capacity for root growth.

A fibrous, well-developed root system is one of the most important seedling attributes that favors regeneration success. To minimize first-year mortality, make sure that total fresh weight (top and roots) is at least 1.4 ounces (4 g). A low

top/root ratio is generally favorable, although this ratio alone is not a reliable predictor of survival.

Seedling morphology can be modified by various nursery techniques. Wrenching—a special form of undercutting can stimulate root growth and retard top growth if done at the proper depth and proper time. This treatment sometimes reduces early mortality of outplanted seedlings.

Careful handling of stock, both at the nursery and at the planting site, will help increase survival. Prolonged root exposure to high temperature and low humidity is especially harmful and should be avoided. Such exposure delays bud-bursting and decreases height growth, needle length, needle weight, terminal bud length, and survival.

On harsh south or southwesterly aspects, shade sometimes improves survival of planted seedlings. Dead shade (from rocks, logs, shingles, etc.) is best, but live shade (from brush or a partial overstory of trees) may also improve survival. However, good survival is possible without shade, even on south aspects, if careful attention is given to all phases of the planting operation. Once established, seedlings nearly always grow best where shade is least.

On steep slopes, soil or debris movement may reduce survival of seedlings by bending them down or burying them. Therefore, plant large seedlings, and carefully choose planting spots on the flattest available microsites, or on the downslope side of stumps or well-anchored rocks.

Container-grown seedlings can be useful in the overall planting program when planted on appropriate sites—principally those characterized by stable soils, and without heavy brush competition or serious animal damage problems. On severe sites the larger bare-root stock generally survives better because of its greater height and diameter, greater proportion of woody tissue (less palatable to animals), and larger root system. On more favorable sites, container-grown stock performs well, and because of the inherent advantage of an intact and relatively undisturbed root system, it can extend the normal planting season moderately.

Vegetative Competition

On some sites, grass can compete seriously with Douglas-fir seedlings soon after they are planted. Atrazine and Terbacil are two chemicals that effectively control grass with little or no damage to Douglas-fir. Dalapon also effectively controls grass when combined with atrazine and 2,4-D, but can injure Douglas-fir when used alone.

Various species of brush and hardwoods provide serious competition for young Douglas-fir trees. The suppressing effects of tree-form hardwoods—such as tanoak, madrone, and red alder, as well as of the taller, denser shrubs such as snowbrush and varnishleaf ceanothus—continue for many years. Species that sprout when cut, burned, or otherwise injured, present a special challenge because of their typically rapid growth rates.

For shade intensities of up to 75 percent, the principal effect of competing vegetation is to reduce vigor and height growth of young Douglas-fir. When shade exceeds this intensity, tree survival also is reduced. Height growth can be reduced as much as 7 feet (2.1 m) during a 10-year period of suppression by competitors such as tanoak and snowbrush.

To control most hardwood and shrub competitors of Douglas-fir, apply low-volatile esters of phenoxy herbicides—2,4-D or 2,4,5-T. When application is properly timed, economical, effective brush control is possible with little or no damage to Douglas-fir. (The 2,4,5-T form is currently banned by the U.S. Environmental Protection Agency, and no fully satisfactory substitute has been found as yet.)

Animal Damage

On a regional basis, the most serious animal damage problem is deer browsing. Numerous approaches have been devised to prevent or reduce browsing, including the use of mechanical barriers, chemical repellents, large planting stock, habitat modification, and genetic modification. Mechanical barriers include devices such as high fences around entire plantations, poultry netting around individual trees, and polyethylene sleeves that protect only the terminal leader. All are effective as long as they remain intact and in place.

Chemical repellents include materials formulated from a variety of substances whose odor or taste are repugnant to wildlife. They are usually less costly than mechanical barriers, but also less effective in terms of duration of protection (repellents deteriorate with weathering, and new growth has no chemical sheath).

Use of large planting stock has received less attention as a means of reducing the impact of animal damage, but it appears to be useful. Large stock is more expensive, however, both to produce and to plant, than is stock of normal size.

Habitat modification and genetic modification of Douglas-fir are still largely in the experimental stages. Both may ultimately become useful methods for reducing deer damage to young trees.

Fertilization, especially with nitrogen, seems to increase palatability of Douglas-fir to deer and, therefore, the likelihood of browsing. The increased susceptibility of Douglas-fir seedlings to browsing extends at least into the second year after fertilization.

After browsed trees finally grow above the reach of deer, they tend to catch up in height to their unbrowsed neighbors. Consequently, growth loss caused by browsing may be insignificant over a rotation. If this proves to be true, efforts to prevent browsing damage may be unnecessary unless tree survival is endangered.

Summary

- Increase seed production in young stands by (a) applying nitrogen and phosphorus, each at the rate of 200 pounds per acre (224 kg/ha); and by (b) stem girdling.
- Prepare sites for regeneration by mechanical or chemical treatment, prescribed burning, or a combination of such treatments.
- If natural regeneration is attempted, schedule harvesting—if possible—to coincide with a good seed year.
- If using direct seeding, follow generally accepted practices.
- Attend carefully to all details during each stage of the planting process—from growing trees from proper seed sources to planting in the field under the most suitable conditions.
- Control vegetative competition to seedlings and young trees with approved herbicides.
- Control damage by animals, chiefly deer, with mechanical barriers or chemical repellents; or use large planting stock.

APPENDIX

Plant and Animal Names

Australian fireweed	*Erechtites prenanthoides* DC.
Bigleaf maple	*Acer macrophyllum* Pursh
Black bear	*Ursus americanus*
Black-tailed deer; also mule deer	*Odocoileus hemionus*
Blueblossom	*Ceanothus thyrsiflorus* Eschsch.
Brown mustard	*Brassica juncea* (L.) Cosson.
Brush rabbit	*Sylvilagus bachmani*
California hazel	*Corylus cornuta* var. *californica* (A.DC.) Sharp
California honeysuckle	*Lonicera hispidula* var. *californica* (Dougl.) Jepson
California red fir	*Abies magnifica* A. Murr.
Canyon live oak	*Quercus chrysolepis* Liebm.
Cone moth or fir coneworm	*Dioryctria abietella* Dennis and Schiffermüller
Cone scale midge	*Contarinia washingtonensis* Johnson
Deerbrush ceanothus	*Ceanothus integerrimus* H. & A.
Deer mouse	*Peromyscus maniculatus*
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*
Douglas-fir cone midge	*Contarinia oregonensis* Foote
Douglas-fir cone moth	*Barbara colfaxiana* Kearf.
Douglas-fir seed chalcid	*Megastigmus spermotrophus* Wachtl
Dusky-footed woodrat	*Neotoma fuscipes*
Elk	*Cervus canadensis*
Fox sparrow	*Passerella iliaca*
Giant chinkapin	*Castanopsis chrysophylla* (Dougl.) A.DC.
Golden-crowned sparrow	*Zonotrichia coronata*
Golden evergreen-chinkapin	*Castanopsis chrysophylla* var. *minor* Benth.
Greenleaf manzanita	*Arctostaphylos patula* Greene
Hairy manzanita	*Arctostaphylos columbiana* Piper
Hoary manzanita	*Arctostaphylos canescens* Eastw.
Howell manzanita	*Arctostaphylos hispidula* Howell
Incense-cedar	*Libocedrus decurrens* Torr.
Jeffrey pine	*Pinus jeffreyi* Grev. & Balf.
Jumping mouse	*Zapus trinotatus*
Kangaroo rat	*Dipodomys* sp.
Knobcone pine	*Pinus attenuata* Lemm.
Meadow mouse	*Microtus* sp.
Mountain beaver	*Aplodontia rufa*
Mountain hemlock	*Tsuga mertensiana* (Bong.) Carr.
Mountain quail	*Oreortyx picta*
Mountain whitethorn ceanothus	*Ceanothus cordulatus* Kell.
Narrow-leaved buckbrush	*Ceanothus cuneatus* (Hook.) Nutt.
New Zealand fireweed	*Erechtites arguta* DC.
Oceanspray	*Holodiscus discolor* (Pursh) Maxim.
Oregon creeping vole	*Microtus oregoni*
Oregongrape	*Berberis nervosa* Pursh
Oregon junco	*Junco oreganus*
Oregon white oak	*Quercus garryana* Dougl. ex Hook.
Pacific dogwood	*Cornus nutallii* Audubon
Pacific madrone	*Arbutus menziesii* Pursh
Pacific red elder	*Sambucus callicarpa* Greene
Pacific rhododendron	*Rhododendron macrophyllum* D. Don ex G. Don
Pocket gopher	*Thomomys* sp.
Poison oak	*Rhus diversiloba* T. & G.
Ponderosa pine	*Pinus ponderosa* Dougl. ex Laws
Port-Orford-cedar	*Chamaecyparis lawsoniana* (A.Murr.) Parl.
Red alder	*Alnus rubra* Bong.
Rose	*Rosa* sp.
Salmonberry	*Rubus spectabilis* Pursh
Scrub tanoak	*Lithocarpus densiflorus* var. *echinoides* (R. Br.) Abrams
Snowbrush ceanothus	*Ceanothus velutinus* Dougl.
Spotted towhee	*Pipilo maculatus*
Squirrel	*Sciurus* sp.
Sugar pine	*Pinus lambertiana* Dougl.
Tanoak	*Lithocarpus densiflorus* (Hook. & Arn.) Rehd.
Townsend's chipmunk	*Eutamias townsendii*
Trowbridge's shrew	*Sorex trowbridgii*
Vagrant shrew	*Sorex vagrans*
Varied thrush	*Ixoreus naevius*
Varnishleaf ceanothus	*Ceanothus velutinus* var. *laevigatus* (Hook.)T.& G.
Vine maple	*Acer circinatum* Pursh
Western hemlock	*Tsuga heterophylla* (Raf.) Sarg.
Western thimbleberry	*Rubus parviflorus* Nutt.
White fir	*Abies concolor* (Gord. & Glend.) Lindl.
White-footed deer mouse	*Peromyscus maniculatus*
Willow	*Salix* sp.

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Strothmann, R. O.; Roy, Douglass F. **Regeneration of Douglas-fir in the Klamath Mountains Region, California and Oregon.** Gen. Tech. Rep. PSW-81. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1984. 35 p.

Information on the regeneration of Douglas-fir, one of the most valuable timber species in the United States, is summarized, from seed production to care of young stands. General recommendations are given to guide the practitioner. Seed production can be increased by applying fertilizer and by stem girdling. To prepare sites for planting, mechanical, burning, chemical or a combination of treatments should be used. All details during each stage of the planting process should be followed. Steps should be taken to control vegetative competition and animal damage to seedlings and young trees. If direct seeding is used, generally accepted practices should be followed. And if natural regeneration is attempted, harvesting should be scheduled to coincide, if possible, with a good seed year.

Retrieval terms: seed production, site preparation, natural regeneration, artificial regeneration, direct seeding, planting, *Pseudotsuga menziessi* (Mirb.) Franco var. *menziessi*, California, Oregon

United States
Department of
Agriculture

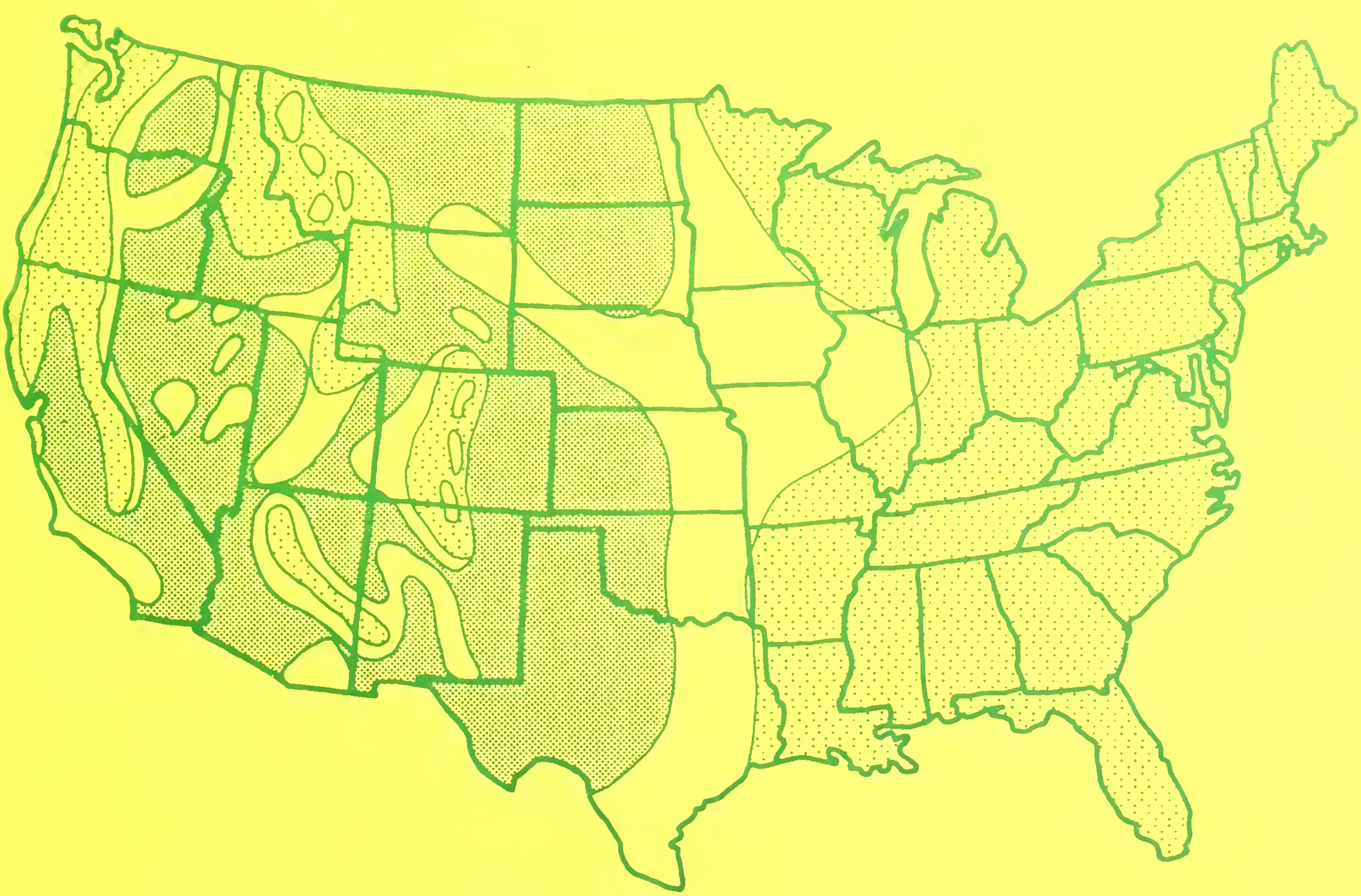
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-82

The National Fire-Danger Rating System: basic equations

Jack D. Cohen John E. Deeming



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Acknowledgments:

The work reported herein was done while we were assigned to the Intermountain Forest and Range Experiment Station's Northern Forest Fire Laboratory at Missoula, Montana. We were aided materially by *Robert E. Burgan*. Other researchers who contributed to the updating of the National Fire-Danger Rating System and their contributions were from the Intermountain Station, Missoula, Montana—*Richard C. Rothermel, Frank A. Albini*, and *Patricia L. Andrews*, who assisted in adapting the current fire modeling technology, including the addition of 1000-hour fuels and herbaceous to 1-hour transfer; *Hal E. Anderson* and *James K. Brown*, who worked on fuels and fuel models; *Donald F. Fuquay*, and *Donald J. Latham*, who worked on the lightning-caused fire occurrence index; North Central Forest Experiment Station, East Lansing, Michigan—*Von J. Johnson, William A. Main*, and *Craig A. Johnson*, who worked on human-caused fire occurrence index; and the Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado—*Michael A. Fosberg*, who worked on fuel moisture models; and *R. William Furman* and *Glen E. Brink*, who collected fire weather data needed to develop the system.

Cover: Four climate classes in the United States are associated with different vegetation moisture contents, which affect fire spread and fire danger rating.

Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

May 1985

The National Fire-Danger Rating System: basic equations

Jack D. Cohen John E. Deeming

CONTENTS

GLOSSARY

AA: Intermediate variable in stick age correction equation.
AD: Exponent in surface area weighted optimum reaction velocity (GMAOP) equation.
ADE: Exponent in loading weighted optimum reaction velocity (GMAOPE) equation.
AGE: Number of days since fuel moisture sticks were set out.
AMBVP: Ambient vapor pressure.
ANNTA: Parameter in linear herbaceous moisture content equation that is used in transition period for annual vegetation.
ANNTB: Parameter in linear herbaceous moisture content equation that is used in transition period for annual vegetation.
ATAN: Trigonometric inverse tangent function of ( ).
B: Wind effect exponent in PHIWND equation.
BB: Intermediate variable in stick age correction equation.
BDYBAR: Seven-day running average of BNDRYT values for calculating MC1000.
BETBAR: Packing ratio.
BETOP: Optimum packing ratio, surface area weighted.
BETOPE: Optimum packing ratio, loading weighted.
BI: NFDRS Burning Index.
BNDRY1: Average boundary moisture condition of first 16 hours of 24-hour forecast period. Applies to predicted 10-hour timelag moisture content.
BNDRY2: Average boundary moisture condition of last 8 hours of 24-hour forecast period. Applies to predicted 10-hour timelag moisture content.
BNDRYH: Weighted 24-hour average moisture condition for 100-hour timelag moisture content calculation (MC100).
BNDRYT: Weighted 24-hour average moisture condition for 1000-hour timelag moisture content calculation (MC1000).
C: Intermediate variable in UFACT equation for calculating wind factor (PHIWND).
CC: Intermediate variable in stick age correction equation.
CELS: Temperature in degrees Celsius.
CGRATE: Cloud-to-ground lightning discharge rate.
CHI: Intermediate variable used in ignition probability (P(I)) equation.
CLIMAT: NFDRS climate class.
CORR: Calculated difference between wet bulb saturation vapor pressure (SATVPW) and ambient vapor pressure (AMBVP).
CURED: AFFIRMS user command to model herbaceous condition as cured.
D: Depth of flaming zone (ft).
DAYLIT: Hours between sunrise and sunset.
DECL: Daily solar zenith angle in radians.
DEDRT: Ratio (WTMCD/MXD) in calculation of ETAMD.
DEDRTE: Ratio (WTMCDE/MXD) in calculation of ETAMDE.
DEPTH: Effective fuel-bed depth measured (ft).
DIFF: Twenty-four hour change in MC1000.
DWPT: Dewpoint temperature.
E: Wind effect exponent in UFACT equation used for calculating wind factor (PHIWND).
ELEV: Elevation of the observing station in feet.
EMC: Equilibrium moisture content.
EMCBAR: Average EMC, weighted by hours of day and night.
EMCBR1: EMC calculated using average temperature and relative humidity from first 16 hours of 24-hour forecast period.
EMCBR2: EMC calculated using average temperature and relative humidity from last 8 hours of 24-hour forecast period.
EMCMAX: EMC calculated from minimum temperature (TMPMIN) and maximum relative humidity (RHMAX).
EMCMIN: EMC calculated from maximum temperature (TMPMAX) and minimum relative humidity (RHMIN).
EMCOBS: EMC calculated from observation time temperature (TMPOBS) and relative humidity (RHOBS).
EMCPRM: EMC calculated using temperature and relative humidity at fuel-atmosphere interface (TMPPRM, RHPRM, respectively).
ERC: NFDRS energy release component.
ETAMD: Surface area weighted dead-fuel moisture damping coefficient.
ETAMDE: Loading weighted dead-fuel moisture damping coefficient.
ETAML: Surface area weighted live-fuel moisture damping coefficent.
ETAMLE: Loading weighted live-fuel moisture damping coefficient.
ETASD: Dead-fuel mineral damping coefficient.
ETASL: Live-fuel mineral damping coefficient.
EXP: Exponential function of ( ).
FAHR: Temperature in degrees Fahrenheit.
F1: Proportion of dead-fuel surface area in 1-hour class, used as a weighting factor for ROS calculation.
F10: Proportion of dead-fuel surface area in 10-hour class, used as weighting factor for ROS calculation.
F100: Proportion of dead-fuel surface area in 100-hour class, used as weighting factor for ROS calculation.
F1E: Proportion of dead-fuel loading in 1-hour class, used as weighting factor for ERC calculation.
F10E: Proportion of dead-fuel loading in 10-hour class, used as weighting factor for ERC calculation.
F100E: Proportion of dead-fuel loading in 100-hour class, used as weighting factor for ERC calculation.
F1000E: Proportion of dead-fuel loading in 1000-hour class, used as weighting factor for ERC calculation.
FCTCUR: Fraction of fuel model herbaceous fuel loading transferred to 1-hour fuel class.
FDEAD: Proportion of total surface area in dead-fuel classes.
FDEADE: Proportion of total loading in dead-fuel classes.
FHERB: Proportion of live surface area in herbaceous class.
FHERBE: Proportion of live loading in herbaceous class.
FINSID: Fraction of total corridor (TOTWID) occupied by lightning-rain corridor.
FL: Byram's flame length (ft).
FLI: NFDRS fire-load index.
FLIVE: Proportion of total surface area in live-fuel classes.
FLIVEE: Proportion of total loading in live-fuel classes.
FMF: Moisture content of 1-hour fuels inside rain area.
FOTSID: Fraction of total corridor (TOTWID) occupied by lightning-rain corridor.
FREEZE: AFFIRMS user command to model herbaceous condition as cured and woody condition as dormant.
FWOOD: Proportion of live surface area in woody class.
FWOODE: Proportion of live loading in woody class.
GMAMX: Weighted maximum reaction velocity of surface area.
GMAMXE: Weighted maximum reaction velocity of loading.
GMAOP: Weighted optimum reaction velocity of surface area.
GMAOPE: Weighted optimum reaction velocity of loading.
GREEN: AFFIRMS user command to start modeled process of herbaceous greenup.
GREN: Fraction of greenup period that has elapsed in calculation of MCHERB.
GRNDAY: Number of days elapsed since greenup started.
HD: Specified dead-fuel heat of combustion of fuel model.

HERBGA: Parameter in equation for linear herbaceous moisture content in greenup period or when MCHERB is greater than 120 percent.
HERBGB: Parameter in equation for linear herbaceous moisture content used in greenup period or when MCHERB is greater than 120 percent.
HL: Specified live-fuel heat of combustion of fuel model.
HN1: Heating number of 1-hour class.
HN10: Heating number of 10-hour class.
HN100: Heating number of 100-hour class.
HNHERB: Heating number of herbaceous class.
HNWOOD: Heating number of woody class.
HTSINK: Heat sink term in ROS equation.
I: Byram's fireline intensity (Btu/ft/s).
IC: NFDRS ignition component.
ICBAR: Area-weighted average ignition component for storm corridor.
ICR: Ignition component calculated for inside rain corridor.
IR: Surface area weighted reaction intensity, used for calculating ROS (Spread Component — SC).
IRE: Loading weighted reaction intensity, used for calculating energy release component (ERC).
IRND: Round-off function of ( ).
JDATE: Julian day of year, 1 to 366, derived from month and day.
KELVIN: Temperature in degrees Kelvin.
KTMP: Temperature factor in X1000 calculation.
KWET: Wetting factor in X1000 calculation.
LAL: NFDRS lightning activity level.
LAT: Station latitude in degrees.
LGTDUR: Duration of lightning at a point within affected area.
LIVRT: Ratio (WTMCL/MXL) in calculation of ETAML.
LIVRTE: Ratio (WTMCLE/MXL) in calculation of ETAMLE.
LOI: NFDRS lightning-caused fire occurrence index.
LRISK: NFDRS lightning risk.
LRSF: Lightning-risk scaling factor.
MC: Moisture content, expressed as percent dry weight.
MC1: Calculated 1-hour timelag percent fuel moisture content.
MC10: Percent moisture content of fuel stick (age corrected) or calculated percent moisture content for 10 - hour timelag.
MC100: Percent moisture content for 100 - hour timelag.
MC1000: Percent moisture content for 1000 - hour timelag.
MC10P1: Predicted MC10 as of next morning of forecast period.
MC10P2: Predicted MC10 for observation time of next day.
MCHERB: Calculated herbaceous percent moisture content.
MCHRB1: Maximum of 30 percent of MCHERB calculated day before greenup.
MCHRBP: Potential herbaceous moisture content during greenup period.
MCLFE: Dead-fuel moisture content, weighted by heating number, for calculation of live moisture of extinction (MXL).
MCOI: NFDRS human-caused index of fire occurrence.
MCWOD1: Greater of PREGRN or MCWOOD calculated day before greenup.
MCWODP: Potential moisture content of woody fuel during modeled greenup period.
MCWOOD: Calculated moisture content of twigs and foliage of woody shrubs.
MRISK: NFDRS human-caused risk.
MXD: Specified dead-fuel moisture of extinction for fuel model.
MXL: Calculated live-fuel moisture of extinction.
P(I): Probability that a firebrand will produce a successful fire start in dead, fine fuels.
P(F/I): Probability that ignition will result in a reportable fire.
PDUR1: Predicted rain duration for first 16 hours of 24-hour forecast period.
PDUR2: Predicted rain duration for last 8 hours of 24-hour forecast period.
PERTA: Parameter in equation for linear herbaceous moisture content used in transition period for perennial vegetation.
PERTB: Parameter in equation for linear herbaceous moisture content used in the transition period for perennial vegetation.
PHI: Station latitude in radians.
PHISLP: Multiplier for slope effect in ROS equation.
PHIWND: Multiplier for wind effect in ROS equation.
PM1000: MC1000 calculated for previous 7th day.
PNORM1: PNORM2 PNORM3: Scaling factors in P(I) calculation that cause P(I) to equal 100 when MC1 is 1.5 percent and zero when MC1 is 25 percent.
PPTAMT: Amount of precipitation.
PPTDUR: Duration of precipitation.
PREGRN: Modeled moisture content of woody shrubs when dormant.
QIGN: Calculated heat of ignition used for calculating P(I).
RAIDUR: Duration of rain at a point within lightning-rain corridor.
RH: Relative humidity.
RHMAX: Maximum relative humidity for 24-hours.
RHMIN: Minimum relative humidity for 24-hours.
RHOBAR: Particle density of weighted fuel.
RHOBED: Bulk density of fuel bed.
RHOBS: Relative humidity of afternoon observation time.
RHOD: Particle density of dead-fuel.
RHOL: Particle density of live-fuel.
RHPRM: Relative humidity estimated for fuel-atmosphere interface.
ROS: Forward rate of spread of flaming front (ft/min).
SA1: Surface area of 1-hour class fuels.
SA10: Surface area of 10-hour class fuels.
SA100: Surface area of 100-hour class fuels.
SADEAD: Surface area of dead-fuel classes.
SAHERB: Surface area of herbaceous fuel class.
SALIVE: Surface area of live-fuel classes.
SATVPD: Saturation vapor pressure for dry bulb temperature (TMPOBS).
SATVPN: Saturation vapor pressure for minimum temperature (TMPMIN).
SATVPW: Saturation vapor pressure for wet bulb temperature (TMPWET).
SATVPX: Saturation vapor pressure for maximum temperature (TMPMAX).
SAWOOD: Surface area of woody fuel class.
SC: NFDRS spread component.
SCM: Specified spread component (SC) of fuel model when all fire starts become reportable fires.
SCN: Percent actual SC is to specified SCM (SC/SCM * 100).
SD: Silica-free mineral content of dead fuels.
SG1: Specified surface area-to-volume ratio for 1-hour class of fuel model.
SG10: Specified surface area-to-volume ratio for 10-hour class of fuel model.
SG100: Specified surface area-to-volume ratio for 100-hour class of fuel model.
SGBRD: Characteristic surface area-to-volume ratio of dead fuel, surface area weighted.
SGBRDE: Characteristic surface area-to-volume ratio of dead fuel, loading weighted.
SGBRL: Characteristic surface area-to-volume ratio of live fuel, surface area weighted.

SGBRLE: Characteristic surface area-to-volume ratio of live fuel, loading weighted.

SGBRT: Characteristic surface area-to-volume ratio of fuel bed, surface area weighted.

SGBRTE: Characteristic surface area-to-volume ratio of fuel bed, loading weighted.

SGHERB: Specified surface area-to-volume ratio for herbaceous class of fuel model.

SGWOOD: Specified surface area-to-volume ratio for woody class of fuel model.

SIN: Trigonometric sine function of ( ).

SL: Proportion of silica-free mineral content of live fuels.

SLPFCT: Coefficient in PHISLP equation.

SQRT: Square-root function of ( ).

STD: Proportion of inert mineral content of dead fuels.

STL: Proportion of inert mineral content of live fuels.

STMDIA: Width of corridor affected by rain-lightning.

STMSPD: Translational speed of storm in miles per hour.

TAN: Trigonometric tangent function of ( ).

TAU: Calculated residence time of flaming front (min).

TEMP: Dry-bulb temperature.

TLA: Intermediate value in daylength calculation.

TLA1: Intermediate value in daylength calculation.

TMPMAX: Twenty-four hour maximum dry-bulb temperature.

TMPMIN: Twenty-four hour minimum dry-bulb temperature.

TMPOBS: Dry-bulb temperature of afternoon observation time.

TMPPRM: Temperature estimated for fuel-atmosphere interface.

TMPWET: Wet-bulb temperature.

TOTWID: Total width of lightning-only and lightning-rain corridor.

UFACT: Wind-effect multiplier in PHIWND equation.

W1: Fuel model specified 1-hour class fuel loading.

W10: Fuel model specified 10-hour class fuel loading.

W100: Fuel model specified 100-hour class fuel loading.

W1000: Fuel model specified 1000-hour class fuel loading.

W1N: Net fuel loading of 1-hour class.

W10N: Net fuel loading of 10-hour class.

W100N: Net fuel loading of 100-hour class.

W1P: One-hour fuel loading and transferred herbaceous loading (WHERBC).

WDEADN: Net loading of dead fuels, surface-area weighted.

WDEDNE: Net dead fuel loading.

WETRAT: Rainfall rate set by climate class (CLIMAT).

WHERB: Herbaceous fuel loading of fuel model.

WHERBC: Amount of herbaceous loading transferred to 1-hour class.

WHERBN: Net fuel loading of herbaceous class.

WHERBP: Amount of herbaceous loading left after transfer to 1-hour fuel loading.

WLIVEN: Surface area weighted net loading of live fuels.

WLIVNE: Net live-fuel loading.

WNDFC: Fuel model specified wind reduction factor. WNDFC is calculated under certain conditions.

WOODGA: Parameter in linear woody moisture content equation used for calculating MCWODP and MCWOOD.

WOODGB: Parameter in linear woody moisture content equation used for calculating MCWODP and MCWOOD.

WRAT: Ratio of dead-to-live heating numbers for calculation of live moisture of extinction (MXL).

WS: Ten-minute average 20 ft windspeed for 10 minutes in miles per hour.

WT: Weight of fuel sticks in grams.

WTMCD: Surface area weighted dead-fuel moisture content.

WTMCDE: Loading weighted dead-fuel moisture content.

WTMCL: Surface area weighted live-fuel moisture content.

WTMCLE: Loading weighted live-fuel moisture content.

WTOT: Total fuel loading.

WTOTD: Total dead-fuel loading.

WTOTL: Total live-fuel loading.

WWOOD: Specified woody shrub loading fuel model.

WWOODN: Net fuel loading of woody class.

X1000: Independent variable in herbaceous fuel moisture models.

YLOI: Previous day's LOI.

YM1000: MC1000 calculated previous day.

YMC10: Initial or current value of 10-hour timelag moisture content for calculating MC10P1.

YMC100: MC100 value calculated previous day.

YX1000: X1000 calculated previous day.

ZETA: No-wind propagating flux ratio in rate-of-spread (ROS) calculation.

The National Fire-Danger Rating System (NFDRS) provides indexes for measuring fire potential in wildlands. It is used by all Federal and many State natural resource management agencies. Data from fire-danger rating stations throughout the United States are processed through the interactive, time-share computer program AFFIRMS (Helfman and others 1980) or are computed manually each day. Fire weather stations record data on the 10-day fire weather record (NWS Form D-9b).

The concept of fire-danger rating and various methods of rating fire-danger have been around for decades. The analytical approach that is the basis for the current NFDRS began in 1968 with the establishment of a National Fire-Danger Rating research work unit. After 4 years of development and field trials, the 1972 NFDRS (Deeming and others 1972) became operational. A review and update was planned at that time.

In 1974, the Chief of the Forest Service, U.S. Department of Agriculture, chartered a technical committee to direct the updating of the NFDRS. A research work unit was chartered and actual work started in 1975. The update work was completed in November 1977, with implementation in May 1978. The results were reported in two publications: one describes the basic system and revisions (Deeming and others 1977); the other explains how to calculate manually fire-danger ratings using the NFDRS (Burgan and others 1977). A third publication summarizes the technical development of the NFDRS (Bradshaw and others 1983).

This report documents the mathematical equations required to calculate fire-danger indexes in the National Fire-Danger Rating System. The equations are in the coded format of FORTAN and BASIC computer languages. They are described in the order processed in the computer program AFFIRMS (Helfman and others 1980) and FIREFAMILY (Main and others 1982), except for equations used to calculate equilibrium moisture content.

CALCULATING MOISTURE CONTENT

The equilibrium moisture content (EMC) is fundamental to all fuel moisture computations in the NFDRS. The EMC, itself a computed value, represents a steady state moisture content of dead woody material. This steady state is achieved under constant conditions for a sufficiently long adjustment period. Steady-state conditions do not occur under normal circumstances and, therefore, do not represent the woody moisture contents. The EMC offers the basis for calculating the various moisture contents considered by the NFDRS.

Equilibrium Moisture Content

Equilibrium moisture contents can be derived from dry bulb temperature and relative humidity by calculating the equilibrium moisture content (EMC).

The following equations for EMC are the regression equations developed by Simard (1968) on the basis of tables in the *Wood Handbook* (U.S. Forest Products Laboratory 1955, revised 1974). Temperatures are expressed in degrees Fahrenheit, and the EMC is expressed as percent moisture content. All variables are explained in the glossary.

Relative Humidity Less Than 10 Percent:

```
EMC = 0.03229 + 0.281073
        * RH - 0.000578 * TEMP * RH        (1a)
```

in which
RH is a relative humidity.
TEMP is a dry bulb temperature.

Relative Humidity Equal to or Greater Than 10 Percent but Less Than 50 Percent:

```
EMC = 2.22749 + 0.160107
        * RH - 0.014784 * TEMP             (1b)
```

Relative Humidity Equal to or Greater Than 50 Percent:

```
EMC = 21.0606 + 0.005565 * RH ** 2
   - 0.00035 * RH * TEMP - 0.483199 * RH   (1c)
```

With these equations, the EMC's can be evaluated for (1) observation time, (2) the time of maximum temperature–minimum relative humidity, and (3) the time of minimum temperature–maximum relative humidity:

```
EMCOBS = f(TMPOBS, RHOBS)
EMCMIN = f(TMPMAX, RHMIN)
EMCMAX = f(TMPMIN, RHMAX)
```

in which

TMPOBS is the dry bulb temperature at the afternoon observation time.

TMPMIN is the 24-hour minimum dry bulb temperature.

TMPMAX is the 24-hour maximum dry bulb temperature.

RHOBS is the relative humidity at the afternoon observation time.

RHMIN is the 24-hour minimum relative humidity.

RHMAX is the 24-hour maximum relative humidity.

Environmental Parameters

Data Available

Calculating fuel moisture contents requires a set of environmental moisture and temperature variables that must be measured or derived. The variables are these:

- Dry bulb temperature and relative humidity at the afternoon observation time;
- Maximum and minimum dry bulb temperatures and relative humidities for the 24-hour period ending at the afternoon observation time; and
- Duration of precipitation during this same 24-hour period.

Operationally, the first and third items are required; measurements to obtain the second item are optional. Historical fire-weather records before 1972 typically do not include the second and third items, although they were often collected for 1972 or a later date. These parameters are calculated, derived, or assigned values.

The dry bulb temperature (TMPOBS) is observed directly and reported in degrees Fahrenheit or degrees Celsius. The relative humidity (RHOBS) may be reported directly, or derived from the dry and wet bulb temperature (TMPOBS and TMPWET) or from the dry bulb temperature and dewpoint (TMPOBS and DWPT). The psychometric equations used for these operations are from the Smithsonian Meteorological Tables (1949). For these calculations, the temperatures are converted to degrees Kelvin by these equations:

KELVIN = CELS + 273.16
KELVIN = (FAHR + 459.69) * 5/9

in which

KELVIN is the temperature in degrees Kelvin.
FAHR is the temperature in degrees Fahrenheit.
CELS is the temperature in degrees Celsius

The Moisture Variable Is Dewpoint Temperature:

RHOBS = 100.0
 * EXP(−7482.6 / (DWPT + 398.36) + 15.674) /
 EXP(−7482.6 / (TMPOBS + 398.36) + 15.674)

The Moisture Variable Is Wet Bulb Temperature: In this situation, the calculation is a bit more complicated. First, calculate the saturation vapor pressures for the wet and dry bulb temperatures (SATVPW and SATVPD).

(dry bulb) SATVPD = EXP(1.81 +
 (TMPOBS * 17.27 − 4717.31) / (TMPOBS − 35.86))
(wet bulb) SATVPW = EXP(1.81 + (TMPWET
 * 17.27 − 4717.31) / (TMPWET − 35.86))

Now calculate an intermediate variable (CORR) to correct for station elevation.

CORR = 6.6 * 10**−4 * (1.0 + (0.00115
 * (TMPWET − 273.16)) * TMPOBS − TMPWET)
 * (1013.09 / EXP(ELEV / 25,000.0))

in which ELEV is the elevation of the observing station in feet. For stations in Alaska, add 200 ft to the actual station elevation to adjust for generally lower surface atmospheric pressures.

Now, the ambient vapor pressure in millibars (AMBVP):

AMBVP = SATVPW − CORR

Relative humidity in percent:

RHOBS = 100.0 * (AMBVP / SATVPD) (2)

Data Unavailable

Relative Humidity—The most common situation observed is a report that includes TMPOBS, a humidity variable (DWPT, RHOBS or TMPWET), and maximum and minimum temperatures (TMPMAX and TMPMIN). Unfortunately, the 24-hour extreme relative humidity data were not collected and, therefore, are not available from pre-1972 reports, and are often not available from post-1972 reports.

When the 24-hour extreme temperatures are reported but the relative humidities are not, it is assumed that the specific humidity at observation time was conserved for the preceding 24-hour period.

Ambient Vapor Pressure:
Rearranging equation (2):

AMBVP = (RHOBS * SATVPD) / 100.0 (3)

Saturation Vapor Pressure for the 24-Hour Maximum and Minimum Temperatures:

(max temp) SATVPX = EXP(1.81 + (TMPMAX
 * 17.27 − 4717.31)
 / (TMPMAX − 35.86))

(min temp) SATVPN = EXP(1.81 + (TMPMIN
* 17.27 - 4717.31)
/ (TMPMIN - 35.86))

24-Hour Maximum and Minimum Relative Humidities:

RHMAX = 100.0 * (AMBVP / SATVPN)
RHMIN = 100.0 * (AMBVP / SATVPX)

Relative Humidity and Temperature— When neither temperature nor relative humidity 24-hour extremes are reported, we estimate EMCMAX and EMCMIN directly. (These data are needed only to obtain values of EMCMAX and EMCMIN, the 24-hour extremes of EMC.) First, we assume that:

EMCMIN = EMCOBS

EMCMAX is assigned a value according to the following scheme:

NFDRS climate class:	Defaulted EMCMAX
1 or 2	15 percent
3 or 4	23 percent

These values were judged to be typical of nighttime recovery values for moderately severe fire-danger situations.

Precipitation— In pre-1972 fire-weather reports, precipitation duration (PPTDUR) was not reported, but precipitation amount (PPTAMT) was. So by assuming a rainfall rate (WETRAT), a pseudo-duration can be calculated as follows:

PPTDUR = IRND((PPTAMT / WETRAT) + 0.49)

in which

IRND indicates that the quantity in parentheses is a rounded integer.

PPTDUR cannot be greater than 8 hours.

WETRAT is a function of climate class as follows:

NFDRS climate class:	WETRAT
	Inches/hour
1 or 2	0.25
3 or 4	0.05

One-quarter inch per hour was judged typical of dry areas (NFDRS climate classes 1 and 2) where the precipitation during the fire season comes usually in the form of moderately intense rain showers. In wetter areas, climate classes 3 and 4, the precipitation is typically much lighter and more or less continuous. PPTDUR, if estimated by this method, is not allowed to exceed 8 hours.

Models of Dead-Fuel Moisture

Fire-danger rating considers two major groups of fuels; live and dead. The live fuels are further classified into annual herbaceous, perennial herbaceous, and lesser woody plants (shrubs and young trees). The 1-, 10-, 100-, and 1000-hour timelag classes represent the dead fuels.

The movement of water vapor between dead-fuel elements and the atmosphere is controlled by the vapor pressure gradient that exists between the two mediums. This gradient is proportional to the difference between the moisture content of the fuel element and the EMC corresponding to the temperature and relative humidity of the air in immediate contact with the fuel element. The dead-fuel moisture calculations are those developed and documented in Fosberg and Deeming (1971); Fosberg (1971); and Fosberg and others (1981).

Fuels: 1-Hour Timelag

The response of 1-hour timelag fuels to changes in the environmental conditions is so rapid that only the potential moisture content, which is equivalent to the EMC at the fuel-atmosphere interface, is required. The first task is to estimate the relative humidity and dry bulb temperature of the air in immediate contact with the fuel elements (TMPPRM and RHPRM). The approach, based on work by Byram and Jemison (1943), consists of correcting the dry bulb temperature and relative humidity values existing at instrument height (4.5 ft) according to the intensity of the insolation (amount of sunshine). Time of year or variables affecting insolation other than cloudiness are not considered. The amount of cloudiness is indicated by the state-of-weather code. The temperature correction is *added* (° F); the relative humidity correction is a *multiplier.*

State-of-weather code:	Corrections: Temperature	Relative humidity
	°F	
0	+ 25	* 0.75
1	+ 19	* 0.83
2	+ 12	* 0.92
3	+ 5	* 1.00

Boundary Layer EMC:

From equations (1a), (1b), or (1c)

EMCPRM = f(TMPPRM, RHPRM)

When Fuel Moisture Sticks Are Not Used:

MC1 = 1.03 * EMCPRM

When Fuel Moisture Sticks Are Used:

MC1 = (4.0 * EMCPRM + MC10) / 5.0

in which MC10 is the 10-hour timelag fuel moisture.

This method was developed for the California wildland fire-danger rating system (U.S. Dep. Agric., Forest Serv. 1958, revised 1968).

If It Is Raining at the Afternoon Observation Time:

MC1 = 35.0

Fuels: 10-Hour Timelag

If an Observation Is Being Processed and Fuel Sticks Are Being Used:

MC10 = Fuel Stick Moisture Content (age corrected)

Because fuel sticks lose weight as they weather, a correction to the measured, apparent moisture content is required. The correction for weathering used in the NFDRS is based on work done by Haines and Frost (1978) as modified by Deeming. A linear model that uses the number of days the sticks have been exposed to the elements and the climate class of the station where the sticks are located are the independent variables.

MC10 = AA * CC + BB * CC * (WT - 100.0)

in which

WT is the weight of the fuel sticks in grams.
AA = 0.5 * AGE / 30.0
BB = 1.0 + (0.02 * AGE / 30.0)
CC = CLIMAT / 4.0
AGE is the number of days since the sticks were set out.
CLIMAT is the NFDRS climate class.

If an Observation Is Being Processed and Fuel Sticks Are Not Being Used:

MC10 = 1.28 * EMCPRM

in which EMCPRM is the same value used in the calculation of MC1.

If a Forecast Is Being Processed and Stick Moisture Content Is Predicted Directly: AFFIRMS allows the fire-weather forecaster to predict stick moisture content directly. A method of making such a prediction was not developed as an integral part of the NFDRS. If this practice is to be used, existing methods such as that by Cramer (1961) are suggested.

If a Forecast Is Being Processed and MC10 Is To Be Calculated: In this situation the model is considerably more complex requiring two computational steps on the basis of work by Fosberg (1977). A fire-weather forecast includes predictions of the minimum temperature and maximum relative humidity for the next 24 hours, the state of weather, temperature, and relative humidity at observation time the next day, and precipitation durations for (a) the first 16 hours of the 24-hour period, and (b) the last 8 hours of the 24-hour period.

The model first predicts the MC10 as of 0600 the next morning (MC10P1), with the current day's MC10 as the initial value (YMC10). With MC10P1 as the initial value, it predicts the potential MC10 at observation time the next afternoon (MC10P2).

Average Boundary Values for Periods 1 and 2 are these:

BNDRY1 = ((16.0 - PDUR1) * EMCBR1
  + (2.7 * PDUR1 + 76.0) * PDUR1) / 16.0
BNDRY2 = ((8.0 - PDUR2) * EMCBR2
  + (2.7 * PDUR2 + 76.0) * PDUR2) / 8.0

in which

PDUR1 and PDUR2 are the predicted durations of precipitation, in hours, for periods 1 and 2.
EMCBR1 and EMCBR2 are the EMC values calculated with the average temperatures and average relative humidities predicted for periods 1 and 2. (The temperatures and relative humidities at observation time the current day and predicted for observation time the next day are corrected for insolation before being averaged with the predicted maximum relative humidity and minimum temperature.)

Moisture Content of the 10-Hour Fuels as of the End of Period 1:

MC10P1 = YMC10 - (BNDRY1 - YMC10)
  * (1.0 - 1.1 * EXP(-1.6))

in which YMC10 is the initial value of the 10-hour fuel moisture as of observation time.

Moisture Content of the 10-Hour Fuels as of the End of Period 2:

MC10P2 = MC10P1 - (BNDRY2 - MC10P1)
  * (1.0 - 0.87 * EXP(-0.8))
MC10 = MC10P2

Fuels: 100-Hour Timelag

Because of the slow response of the 100-hour and the 1000-hour classes of fuels to changes in environmental conditions, we use an EMC that represents the average drying-wetting potential of the atmosphere for the preceding 24-hour period. The 24-hour average EMC is denoted as EMCBAR, a weighted average of EMCMAX and EMCMIN. Weighting is done on the basis of hours of daylight and hours of darkness that are functions of latitude and date.

Duration of Daylight:

PHI = LAT * 0.01745

in which

LAT is the station latitude in degrees.
DECL = 0.41008 * SIN((JDATE-82) * 0.01745)

in which

JDATE is the Julian date.

DECL is the solar declination in radians.

DAYLIT = 24 * (1. – ACOS(TAN(PHI) * TAN(DECL))/3.1416)

in which DAYLIT is the number of hours between sunrise and sunset.

Weighted 24-Hour Average EMC:

EMCBAR = (DAYLIT * EMCMIN + (24.0 – DAYLIT) * EMCMAX) / 24.0

Weighted 24-Hour Average Boundary Condition:

BNDRYH = ((24.0 – PPTDUR) * EMCBAR + PPTDUR * (0.5 * PPTDUR + 41.0)) / 24.0

in which PPTDUR is the hours of precipitation reported (predicted) for the 24-hours.

100-Hour Timelag Fuel Moisture: The model used in the manual version of the 1978 NFDRS to calculate the 100-hour timelag fuel moisture differs from this model in two ways: (1) daylength is not considered, and (2) the 24-hour average EMC is a function of the simple averages of the 24-hour temperature and relative humidity extremes.

MC100 = YMC100 + (BNDRYH – YMC100) * (1.0 – 0.87 * EXP(–0.24))

in which YMC100 is the MC100 value calculated the previous day.

Initializing YMC100 at the Beginning of a Computational Period:

YMC100 = 5.0 + (5.0 * CLIMAT)

Fuels: 1000-Hour Timelag

Weighted 24 Hour Average Boundary Condition:

BNDRYT = ((24.0 – PPTDUR) * EMCBAR + PPTDUR * (2.7 * PPTDUR + 76.0)) / 24.0

Seven Day Running Average Boundary Condition:

BDYBAR = (BNDRYT(1) + ... + BNDRYT(7)) / 7.0

in which ( ) denotes a day in the 7-day series. It is necessary, therefore, to maintain a 1 × 7 array of BNDRYT values.

1000-Hour Timelag Fuel Moisture: The model used in the manual version of the 1978 NFDRS to calculate the 1000-hour timelag fuel moisture differs from this model in the following ways: (1) daylength is not considered, (2) the 24-hour average EMC is a function of the simple averages of the 24-hour temperature and relative humidity extremes, and (3) BNDRYT is calculated daily, but BDYBAR and MC1000 are calculated only every seventh day.

MC1000 = PM1000 + (BDYBAR – PM1000) * (1.00 – 0.82 * EXP(–0.168))

in which PM1000 is the MC1000 calculated for the seventh previous day.

It is necessary, therefore, to maintain a 1 × 7 listing of MC1000 values:

BNDRYT and MC1000 array

	1	2	3	4	5	6	7	
X_a	X_b	X_c	X_d	X_e	X_f	X_g	X_h	
			(next day)					
X_b	X_c	X_d	X_e	X_f	X_g	X_h	X_i	

When a new value is added to the BNDRYT or MC1000 (X_h) arrays, the existing values are moved one position down the array. The value for the oldest day of the 7-day series (X_a) is replaced by the value for the next most recent day (X_b) and so on through the entire array. The latest BNDRYT and MC1000 values (X_i) are placed in the seventh position of the arrays.

Initializing the MC1000 and BNDRYT Arrays at the Beginning of a Computational Period:

MC1000 (n) = 10.0 + (5.0 * CLIMAT)
BNDRYT (n) = 10.0 + (5.0 * CLIMAT)

in which (n) denotes cells 1 through 7 in the two arrays.

Fuels: Wet or Ice-Covered

Rather than complicate the component models of the NFDRS with intricate logic, these conditions were judged better dealt with by rules. The wildfire potential is zero when the ground fuels are wet or covered with ice or snow. The expected data must be provided to the computer (AFFIRMS), however, and the 100- and 1000-hour fuel moisture calculations must continue without interruption.

The logic of the rules is not difficult to accept:

- No wildfire potential exists when ice, snow, or both are present.
- Free water affects fuels in essentially the same way, whether it comes in the various forms of precipitation or from the thawing of ice or snow.
- When precipitation is frozen or fuels are covered with ice, snow, or both, and there is no thaw, fuel moistures respond as if the relative humidity were 100 percent (McCammon 1974).

Raining, Snowing, and Thawing, or Fuels Wet (Observation and Forecast):

1. Set SC, ERC, BI, IC, MCOI, LOI, and FLI to Zero (0).
2. Record MC1 as 35 percent.
3. Record MC10 as 35 percent for the predicted value and for the observed value if sticks are not used: if sticks are

used, shake off the water and weigh as usual for the observed value.

4. Calculate the 100- and 1000-hour fuel moistures as usual.

Snowing or Fuels Covered by Ice, Snow, or Both, No Thaw (Observation and Forecast):

1. Set SC, ERC, BI, IC, MCOI, LOI, and FLI to zero (0).
2. Record MC1 as 35 percent.
3. Record MC10 as 35 percent for the predicted value and for the observed value if sticks are not used; if sticks are used, shake off the snow and weigh as usual for the observed value.
4. Calculate the 100- and 1000-hour fuel moistures as usual except use zero (0) for PPTDUR and 100 percent for both RHMAX and RHMIN.

Fuels Covered by Ice, Snow, and Thawing:

1. Set SC, ERC, BI, IC, MCOI, LOI, and FLI to zero (0).
2. Record MC1 as 35 percent.
3. Record MC10 as 35 percent for the predicted value and for the observed value if sticks are not used: if sticks are used, shake off the snow and weigh as usual for the observed value.
4. Calculate the 100- and 1000-hour fuel moistures as usual except use the duration of the thaw during that 24-hour period for PPTDUR and 100 percent for both RHMAX and RHMIN.

Models of Live-Fuel Moisture

For a more complete description of the live-fuel moisture models, refer to the publication by Burgan (1979).

Essentially, two live-fuel moisture models are contained in the NFDRS:

1. Herbaceous fuel moisture model with variations for annual and perennial types;
2. Woody shrub fuel moisture model.

Herbaceous Fuels

The loading of the herbaceous fuels is a fuel model parameter just as the 1-hour timelag fuel loading is a fuel model parameter. The user specifies the herbaceous type as annual or perennial and the NFDRS climate class of the observation site. Those parameters control the rate at which the model passes through these five stages:

1. pregreen (MCHERB 30 percent or less)
2. greenup
3. green (MCHERB greater than 120 percent)
4. transition (MCHERB 30 to 120 percent)
5. cured or frozen (MCHERB 30 percent or less)

in which MCHERB is the moisture content of the herbaceous plants.

In late summer or fall herbaceous plants die or are killed back by frost. They remain, in either state, waiting for the warmth and moisture of spring before starting another growth cycle. When the herbaceous plants are in the *pregreen* stage, the model assumes their moisture content responds to the environment the same as the moisture content of the 1-hour fuels. The model accounts for this by transferring the entire loading of herbaceous plants to the 1-hour fuel class.

When greenup is declared by the user in spring, the herbaceous moisture content is initially at 30 percent. The loading of the herbaceous fuel is then reclaimed from the 1-hour dead-fuel class as the herbaceous moisture content increases to 120 percent. The length of the greenup period depends on the NFDRS climate class of the observation site being 1 week for class 1, 2 weeks for class 2, 3 weeks for class 3, and 4 weeks for class 4.

If greenup is complete, the MCHERB is 120 percent, the model has reached the *green* stage. The moisture content of perennials is allowed to increase or decrease as the available moisture increases or decreases. The moisture content of annuals can only remain the same or decrease once the greenup process is complete. The maximum value of MCHERB is 250 percent.

When MCHERB falls below 120 percent, the herbaceous vegetation enters *transition;* that is, the curing process begins. As MCHERB decreases below 120 percent, the herbaceous loading is transferred to the 1-hour fuel class—the reverse of what occurred during greenup. When MCHERB falls to 30 percent, the loading of herbaceous fuel is zero. During transition, perennial plants are allowed to pick up moisture, but annual plants are not.

If MCHERB decreases to 30 percent, the plants are considered *cured.* Perennial plants are allowed to recover and "re-green" on their own if moisture conditions improve. Once cured, however, annuals stay cured until the user declares "GREEN."

The user can force the model to cure the herbaceous plants by specifying either "CURED" or "FROZEN." The herbaceous moisture prediction model responds identically to these two declarations. ("FROZEN" forces shrubs into dormancy, however). In this situation, both annuals and perennials remain cured until the next greenup is declared.

Equations for the model through the pregreen, greenup, green, transition, and cured-frozen stages are these:

Pregreen Stage—This stage is best described as that existing in late winter before new growth appears. The herbaceous plants are "straw-brown," having been killed by low moisture or freezing temperatures the previous year. The pregreen stage is set up in AFFIRMS only if a "FROZEN" command has been entered or if there has been a break in the data of at least 60 days. In milder climates, such as the Southeast, Gulf Coast, or Southwest, a killing freeze may not occur. So in those areas, if AFFIRMS is run year-long,

the pregreen state should be forced during the winter with a "FROZEN" command.

The following model settings are made for the pregreen stage:

MCHERB = MC1
W1P = W1 + WHERB

in which

W1 is the fuel model 1-hour fuel loading.
WHERB is the fuel model herbaceous fuel loading.
W1P is the pregreen 1-hour fuel loading.

Greenup Stage—When the greenup process is begun the model settings are these:

MCHERB = MCHRBI
W1P = W1 + WHERB
X1000 = MC1000
GRNDAY = 0

in which

MCHRBI is the maximum of 30 percent or MCHERB calculated the day before to the greenup.
W1P is the pregreen 1-hour fuel loading.
X1000 is the independent variable in the herbaceous fuel moisture models
GRNDAY is the number of days elapsed since greenup started.

MCHERB is a function of X1000, the herbaceous plant type, and the NFDRS climate class. With the start of greenup, X1000 is set equal to MC1000. From that point on, the X1000 is calculated as follows:

DIFF = MC1000 - YM1000
X1000 = YX1000 + (DIFF * KWET * KTMP)

in which

YM1000 is the MC1000 calculated the previous day.
YX1000 is the X1000 calculated the previous day.
DIFF is the 24-hour change in MC1000.
KWET is the wetting factor.
KTMP is the temperature factor.

in which

If MC1000 is greater than 25 percent, KWET = 1.0.
If MC1000 is less than 26 percent and greater than 9 percent, KWET = (0.0333 * MC1000 + 0.1675).
If MC1000 is less than 10 percent, KWET = 0.5.
If DIFF is less than or equal to 0.0, KWET = 1.0.
If (TMPMAX + TMPMIN) / 2.0 is less than or equal to 50°F, KTMP = 0.6; otherwise, a value of 1.0 is used.

Next needed is the moisture content that herbaceous fuels would have *if the greenup period were over;* this we call MCHRBP (P for Potential). MCHRBP is linearly related to X1000, but the constants of the relationship are functions of the NFDRS climate class.

MCHRBP = HERBGA + HERBGB * X1000

in which the constant and coefficient are determined by the NFDRS climate class.

NFDRS Climate class:	HERBGA	HERBGB
1	- 70.0	12.8
2	-100.0	14.0
3	-137.5	15.5
4	-185.0	17.4

The length of the greenup period, in days, is seven times the NFDRS climate class. The fraction of greenup period that has elapsed must be calculated so that the loading of the herbaceous fuel can be calculated.

GREN = GRNDAY / (7.0 * CLIMAT) (4)

in which GRNDAY is the number of days since the greenup sequence was started.

During greenup, MCHERB is phased up.

MCHERB = MCHRBI + (MCHRBP - MCHRBI) * GREN

The fraction of the fuel model herbaceous fuel loading transferred to the 1-hour class is FCTCUR.

FCTCUR = 1.33 - 0.0111 * MCHERB (5)

in which FCTCUR cannot be less than 0.0 nor more than 1.0.

The actual amount of fuel transferred can now be calculated.

WHERBC = FCTCUR * WHERB (6)

The 1-hour and herbaceous fuel loadings, W1P and WHERBP, therefore, become:

W1P = W1 + WHERBC (7)

WHERBP = WHERB - WHERBC (8)

Green Stage—At the end of the greenup period GREN = 1.0; therefore, MCHERB = MCHRBP. If it has been exceptionally dry during the greenup period, X1000 will be low, and MCHRBP may not reach 120 percent. When this situation occurs, the green stage is bypassed and the model goes directly into transition. As long as MCHRBC is greater than 120 percent, however, MCHERB (for both perennials and annuals) is calculated by the linear equation and constants determined by the climate class (see tabulation of HERBGA AND HERBGB above).

MCHERB = HERBGA + HERBGB * X1000

in which MCHERB is not allowed to exceed 250 percent.

Transition Stage—When MCHERB drops below 120 percent, these transition equations are used:

For annuals:

MCHERB = ANNTA + ANNTB * X1000

For perennials:

MCHERB = PERTA + PERTB * X1000

in which

MCHERB cannot exceed 150 or be less than 30 percent if plants are perennials.

MCHERB cannot be higher than MCHERB calculated the previous day if plants are annuals.

The constant and coefficient for the equations are determined by the NFDRS climate class as follows:

NFDRS	Annuals		Perennials	
climate class:	ANNTA	ANNTB	PERTA	PERTB
1	−150.5	18.4	11.2	7.4
2	−187.7	19.6	−10.3	8.3
3	−245.2	22.0	−42.7	9.8
4	−305.2	24.3	−93.5	12.2

For both herbaceous types, the model causes fuel to be transferred back and forth between the herbaceous and 1-hour classes as MCHERB fluctuates between 30 and 120 percent. Equations 5, 6, 7, and 8 are used to calculate the amount of fuel moving back and forth between classes.

Cured or Frozen Stage—If curing occurs by way of normal, seasonal drying (MCHERB drops to 30 percent without intervention by the user), these equations are used:

For annuals:

MCHERB = MC1

For perennials:

MCHERB = PERTA + PERTB * X1000

in which MCHERB cannot be less than 30 percent nor more than 150 percent if the plants are perennials.

Perennials are treated no differently than they are in the transition stage unless the user forces curing with a FROZEN or CURED command. If that is the situation, the equation for MCHERB for perennials becomes:

MCHERB = MC1

In automated systems such as AFFIRMS and FIREFAMILY, once curing has taken place, the herbaceous moisture content of annual plants is displayed with the same value as the moisture content of the 1-hour fuels. If the perennial designation has been used, the herbaceous moisture content will remain at 30 percent until moisture conditions improve or FROZEN or CURED has been declared. Operationally, a killing freeze is declared by the user. With historical fire-weather data, FIREFAMILY handles freezing as follows: the user designates on a lead card the earliest date that a killing frost is plausible. FIREFAMILY, for observations after that date, will then set the FROZEN flag:

1. The first time a minimum temperature (TMPMIN) 25° F or less occurs; or
2. The fifth day that a minimum temperature falls in the range 26° to 32° F.

Shrub Fuels

The prediction model for woody fuel moisture is much simpler than that for herbaceous fuel moisture. Only four stages in the annual growth cycle are recognized, and loadings between fuel classes are not transferred. The four stages of the model are:

1. pregreen (MCWOOD = PREGRN)
2. greenup
3. green (MCWOOD greater than PREGRN)
4. frozen (MCWOOD = PREGRN)

in which

MCWOOD is the predicted moisture content of the twigs and foilage of the shrubs.

PREGRN is the moisture content of shrubs when dormant.

Pregreen Stage—This stage is analagous to the pregreen stage of the herbaceous fuel moisture model.

MCWOOD = PREGRN

in which PREGRN is 50 for NFDRS climate class 1, 60 for class 2, 70 for class 3, or 80 percent for class 4.

Greenup Stage—MCWOOD is a function of MC1000 and the NFDRS climate class. During greenup, as in the herbaceous fuel moisture model, MCWOOD increases to its potential value over a period equal to seven times the climate class of the observation site.

The first step is the calculation of the potential woody fuel moisture

MCWODP = WOODGA + WOODGB * MC1000

in which the constant and coefficient are determined by the NFDRS climate class.

NFDRS climate class:	WOODGA	WOODGB
1	12.5	7.5
2	−5.0	8.2
3	−22.5	8.9
4	−45.0	9.8

The fraction of the greenup period that has elapsed, GREN, is calculated by equation 4. The woody fuel moisture during greenup is then calculated by this equation:

MCWOOD = MCWODI
+ (MCWODP − MCWODI) * GREN

in which MCWODI is the greater of PREGRN or the MCWOOD calculated the day previous to greenup.

Green Stage—After greenup, GREN = 1.0 and MCWOOD = MCWODP, and, therefore, MCWOOD can be calculated by the linear equation and constants from the preceding table.

MCWOOD = WOODGA + WOODGB * MC1000

in which MCWOOD cannot be less than the appropriate PREGRN value or greater than 200 percent.

Frozen Stage—This is the same as the pregreen stage, but is included to provide consistency between the discussions of the herbaceous and woody moisture models.

The model for woody fuel moisture does not respond to CURED if and when it is declared by the user. The types of shrubs considered here are not deciduous, therefore, no condition analogous to herbaceous cured exists. Shrubs will go into dormancy if the weather is cold enough, however, so the model for woody fuel moisture responds to FROZEN by setting MCWOOD to the appropriate PREGRN value. MCWOOD will not change until the next greenup.

SYSTEM COMPONENTS AND INDEXES

The NFDRS spread and energy release components (SC and ERC) are taken from Rothermel (1972) as modified by Albini (1976). The burning index (BI) is based on Byram's flame-length model (Byram 1959). The ignition and the two occurrence models were developed for the NFDRS, as was the fire load index (FLI). The SC, ERC, and BI relate to the characteristics of fire. These models will be dealt with first.

Models of Fire Characteristics

Preliminary Calculations

All fuel loadings are converted to pounds per square foot by multiplying the tons per acre value by 0.0459137 (lb-acre)/(ton-ft^2). Fuel model characteristics are listed in the *appendix*.

Total Dead and Live-Fuel Loadings:

(dead) WTOTD = W1P + W10 + W100 + W1000
(live) WTOTL = WHERBP + WWOOD

in which
W1P is the 1-hour timelag fuel loading in addition to the portion of the live herbaceous loading that has cured.
W10, W100, and W1000 are the loadings of the 10-, 100-, and 1000-hour timelag fuels specified in the fuel model.
WHERBP is the herbaceous fuel loading remaining.
WWOOD is the loading of the woody shrub plants specified in the fuel model.

Total Fuel Loading:

WTOT = WTOTD + WTOTL

Net Fuel Loading of Each Fuel Class:

(1-hour) W1N = W1P * (1.0 − STD)
(10-hour) W10N = W10 * (1.0 − STD)
(100-hour) W100N = W100 * (1.0 − STD)
(herbaceous) WHERBN = WHERBP * (1.0 − STL)
(woody) WWOODN = WWOOD * (1.0 − STL)

in which STD and STL are the fractions of the dead and live fuels made up of inert, noncombustible minerals. A value of 0.0555 is used for both.

Bulk Density of the Fuel Bed:

RHOBED = (WTOT − W1000) / DEPTH

in which DEPTH is the effective fuel bed depth measured in feet.

Weighted Fuel Density:

RHOBAR = ((WTOTL * RHOL)
+ (WTOTD * RHOD)) / WTOT

in which
RHOL and RHOD are the live and dead-fuel particle densities.
A constant value of 32 lb/ft^3 is used.
Packing Ratio:

BETBAR = RHOBED/RHOBAR

Mineral Damping Coefficient of Live and Dead Fuels:

(dead) ETASD = 0.174 * SD**(−0.19)
(live) ETASL = 0.174 * SL**(−0.19)

in which SD and SL are the fractions of the dead and live fuels made up of silica-free, noncombustible minerals. A constant value of 0.01 is assumed for both SD and SL.

Heating Numbers of Each Fuel Class:

```
    (1-hour)   HN1 = W1N * EXP(-138.0 / SG1)
   (10-hour)   HN10 = W10N * EXP(-138.0 / SG10)
  (100-hour)   HN100 = W100N
                          * EXP(-138.0 / SG100)
(herbaceous)   HNHERB = WHERBN
                          * EXP(-500.0 / SGHERB)
     (woody)   HNWOOD = WWOODN
                          * EXP(-500.0 / SGWOOD)
```

in which SG1, SG10, SG100, SGHERB, and SGWOOD are the surface-area-to-volume ratios of the 1-, 10-, 100-hour herbaceous and woody fuels specified in the fuel model.

Because the surface-area-to-volume ratio of the 1000-hour fuel class is so low (8.0 ft^{-1}), its influence is minimal and is omitted to simplify the computation. No net 1000-hour fuel loading is computed for the same reason.

Ratio of Dead-to-Live Fuel Heating Numbers:

```
WRAT = (HN1 + HN10 + HN100)
                / (HNHERB + HNWOOD)
```

Spread Component

In this model, the influence each fuel class has on the result is determined by the fraction of the total *surface area* of the fuel complex contributed by that fuel class.

Surface Area of Each Fuel Class:

```
    (1-hour)   SA1 = (W1P/RHOD) * SG1
   (10-hour)   SA10 = (W10/RHOD) * SG10
  (100-hour)   SA100 = (W100/RHOD) * SG100
(herbaceous)   SAHERB = (WHERB/RHOL)
                          * SGHERB
     (woody)   SAWOOD = (WWOOD/RHOL)
                          * SGWOOD
```

Total Surface Area of Dead and Live Fuels:

```
(dead)   SADEAD = SA1 + SA10 + SA100
(live)   SALIVE = SAHERB
                   + SAWOOD
```

Weighting Factors of Each Fuel Class:

```
    (1-hour)   F1 = SA1/SADEAD
   (10-hour)   F10 = SA10/SADEAD
  (100-hour)   F100 = SA100/SADEAD
(herbaceous)   FHERB = SAHERB/SALIVE
     (woody)   FWOOD = SAWOOD/SALIVE
```

Weighting Factors of Dead and Live Fuels:

```
(dead)   FDEAD = SADEAD/(SADEAD + SALIVE)
(live)   FLIVE = SALIVE/(SADEAD + SALIVE)
```

Weighted Net Loadings of Dead and Live Fuels:

```
(dead)   WDEADN = (F1 * W1N) + (F10 * W10N)
                   + (F100 * W100N)
(live)   WLIVEN = (FWOOD * WWOODN)
                   + (FHERB * WHERBN)
```

Dead and Live Fuel Characteristic Surface-Area-to-Volume Ratios:

```
(dead)   SGBRD = (F1 * SG1) + (F10 * SG10)
                   + (F100 * SG100)
(live)   SGBRL = (FHERB * SGHERB)
                   + (FWOOD * SGWOOD)
```

Characteristic Surface-Area-to-Volume Ratio:

```
SGBRT = (FDEAD * SGBRD) + (FLIVE * SGBRL)
```

Optimum Packing Ratio:

```
BETOP = 3.348 * SGBRT**(-0.8189)
```

Maximum Reaction Velocity:

```
GMAMX = (SGBRT**1.5) / (495.0
                   + 0.0594 * SGBRT**1.5)
```

Optimum Reaction Velocity:

```
GMAOP = GMAMX*(BETBAR/BETOP)**AD
          * EXP(AD * (1.0 - BETBAR / BETOP))
```

in which AD = 133.0 * SGBRT**(-0.7913)

No Wind Propagating Flux Ratio:

```
ZETA = EXP((0.792 + 0.681 * SGBRT**0.5)
        * (BETBAR + 0.1)) / (192.0 + 0.2595 * SGBRT)
```

Weighted Dead-Fuel Moisture Content for Live-Fuel Extinction Moisture:

```
MCLFE = ((MC1 * HN1) + (MC10 * HN10)
        + (MC100 * HN100)) / (HN1 + HN10 + HN100)
```

Moisture of Extinction of Live Fuels:

```
MXL = (2.9 * WRAT * (1.0 - MCLFE/MXD)
                   - 0.226) * 100.0
```

in which

MXD is the moisture of extinction of the dead fuels from the fuel model.
MXL cannot be less than MXD.

Weighted Moisture Content of Dead and Live Fuels:

(dead) WTMCD = (F1 * MC1) + (F10 * MC10) + (F100 * MC100)
(live) WTMCL = (FHERB * MCHERB) + (FWOOD * MCWOOD)

Moisture Damping Coefficients of Dead and Live Fuels:

(dead) ETAMD = 1.0 - 2.59 * DEDRT + 5.11 * DEDRT**2.0 - 3.52 * DEDRT**3.0
(live) ETAML = 1.0 - 2.59 * LIVRT + 5.11 * LIVRT**2.0 - 3.52 * LIVRT**3.0

in which
DEDRT = (WTMCD / MXD)
LIVRT = (WTMCL / MXL)
ETAMD and ETAML cannot be less than zero or greater than 1.0.
Wind Effect Multiplier Coefficients and Exponents:

B = 0.02526 * SGBRT**0.54
C = 7.47 * EXP(-0.133 * SGBRT**0.55)
E = 0.715 * EXP(-3.59 * 10.0**(-4.0) * SGBRT)
UFACT = C * (BETBAR / BETOP)**(-E)

Wind Effect Multiplier:

PHIWND = UFACT * (WS * 88.0 * WNDFC)**B

in which
WS is the 10-minute average 20-ft windspeed in mph.
WNDFC is the fuel model wind reduction factor.
The effect of high winds is limited:
If (WS * 88.0 * WNDFC) is greater than (0.9 * IR), then (0.9 * IR) replaces (WS * 88.0 * WNDFC). Then the equation becomes

PHIWND = UFACT * (0.9 * IR)**B

Slope Effect Multiplier Coefficient:

SLPFCT = 5.275 * (TAN(slope angle))**2.0

in which

NFDRS slope class:	Slope angle	SLPFCT
1	12.67°	0.267
2	17.63°	0.533
3	24.23°	1.068
4	32.46°	2.134
5	41.99°	4.273

Slope Effect Multiplier:

PHISLP = SLPFCT * BETBAR**(-0.3)

Reaction Intensity:

IR = GMAOP * ((WDEADN * HD * ETASD * ETAMD) + (WLIVEN * HL * ETASL * ETAML))

in which HD and HL are the heat values for dead and live fuels specified in the fuel model, Btu/lb.

Heat Sink:

HTSINK = RHOBED * (FDEAD
* (F1 * EXP(-138.0/SG1) * (250.0 + 11.16 * MC1)
+ F10 * EXP(-138.0/SG10) * (250.0 + 11.16 * MC10)
+ F100 * EXP(-138.0/SG100) * (250.0 + 11.16 * MC100)))
+ (FLIVE * (FHERB * EXP(-138.0/SGHERB)
* (250.0 + 11.16 * MCHERB) + FWOOD
* EXP(-138.0/SGWOOD)
* (250.0 + 11.16 * MCWOOD)))

Rate of Spread:

ROS = IR * ZETA * (1.0 + PHISLP + PHIWND)/HTSINK (ft/min)

Spread Component:

SC = IRND(ROS)

Energy Release Component

In this model the influence of each fuel class is determined by the fraction of the total fuel loading contributed by that class. As a result, the conditions of the larger fuels have more influence on the fire-danger.

Weighting Factors of Each Fuel Class:

(1-hour) F1E = W1P/WTOTD
(10-hour) F10E = W10/WTOTD
(100-hour) F100E = W100/WTOTD
(1000-hour) F1000E = W1000/WTOTD
(herbaceous) FHERBE = WHERBP/WTOTL
(woody) FWOODE = WWOOD/WTOTL

Weighting Factors of Dead and Live Fuels:

(dead) FDEADE = WTOTD/WTOT
(live) FLIVEE = WTOTL/WTOT

Net Loadings of Dead and Live Fuels:

(dead) WDEDNE = WTOTD * (1.0-STD)
(live) WLIVNE = WTOTL * (1.0-STL)

Dead and Live Fuel Characteristic Surface-Area-to-Volume Ratios:

(dead) SGBRDE = (F1E * SG1) + (F10E * SG10)
+ (FC100E * SG100)
+ (F1000E * SG1000)
(live) SGBRLE = (FWOODE * SGWOOD)
+ (FHERBE * SGHERB)

Characteristic Surface-Area-to-Volume Ratio:

SGBRTE = (FDEADE * SGBRDE)
+ (FLIVEE * SGBRLE)

Optimum Packing Ratio:

BETOPE = 3.348 * SGBRTE**(-0.8189)

Maximum Reaction Velocity:

GMAMXE = SGBRTE**1.5/(495.0
+ 0.0594 * SGBRTE**1.5)

Optimum Reaction Velocity:

GMAOPE = GMAMXE
* (BETBAR/BETOPE)**ADE*EXP(ADE
* (1.0 - BETBAR/BETOPE))

in which ADE = 133.0 * SGBRTE**(-0.7913)

Weighted Moisture Contents of Dead and Live Fuels:

(dead) WTMCDE = (F1E * MC1) + (F10E * MC10)
+ (F100E * MC100)
+ (F1000E * MC1000)
(live) WTMCLE = (FWOODE * MCWOOD)
+ (FHERBE * MCHERB)

Moisture Damping Coefficients of Dead and Live Fuels:

(dead) ETAMDE = 1.0 - 2.0 * DEDRTE
+ 1.5 * DEDRTE**2.0
- 0.5 * DEDRTE**3.0
(live) ETAMLE = 1.0 - 2.0 * LIVRTE
+ 1.5 * LIVRTE**2.0
- 0.5 * LIVRTE**3.0

in which
DEDRTE = (WTMCDE/MXD)
LIVRTE = (WTMCLE/MXL)
ETAMDE and ETAMLE cannot be less than zero or greater than 1.0

Reaction Intensity:

IRE = GMAOPE * ((FDEADE * WDEDNE * HD
* ETASD * ETAMDE) + (FLIVEE * WLIVNE
* HL * ETASL * ETAMLE))

Residence time of the Flaming Front:

TAU = 384.0/SGBRT

The surface area weighted surface area-to-volume ratio, SGBRT, is used rather than the mass weighted form (SGBRTE). The mass weighted residence time produced unrealistic results.

Energy Release Component:

ERC = IRND(0.04 * IRE * TAU)

The 0.04 scaling factor has the units ft^2/Btu. As such, a unit value of ERC is equivalent to 25 Btu of available energy per square foot.

Burning Index

The BI is numerically equivalent to 10 times the predicted flame length, in feet. The equation developed by Byram (1959) is used with some liberties, enabling us to use parameters that are outputs from Rothermel's fire spread model.

Byram's equation:

FL = 0.45 * I**0.46 (ft.)

in which I is the fireline intensity, Btu/ft-sec
but I = IRE * D/60.0, Btu/ft-sec
in which D = ROS * TAU, ft
so I = (ROS/60.0) * IRE * TAU, Btu/ft-sec
but ROS = SC and IRE * TAU = 25.0 * ERC
therefore FL = 0.45 * ((SC/60.0) * (25.0 * ERC))**0.46
and FL = 0.301 * (SC * ERC)**0.46

Burning Index:

BI = IRND(3.01 * (SC * ERC)**0.46)

If the fuels are wet or covered by snow or ice at observation time, the BI is set to zero.

Models of Fire Occurrence

Ignition Component

The IC consists of two parts: (1) the probability that a firebrand will produce a successful fire start in dead, fine fuels, P(I); and (2) the probability that a reportable fire will occur, given an ignition P(F/I).

P(I) is a function of the amount of heat required to produce an ignition (QIGN) which, in turn, is a function of the 1-hour fuel moisture, MC1. P(I) is scaled such that it is 100 when MC1 = 1.5 percent and zero when MC1 = 25.0 percent. Three scaling factors are used for this purpose:

PNORM1 = 0.00232
PNORM2 = 0.99767
PNORM3 = 0.0000185

Heat of Ignition:

QIGN = 144.5 − (0.266 * TMPPRM) − (0.00058 * TMPPRM**2.0) − (0.01 * TMPPRM * MC1) + (18.54 * (1.0 − EXP(−0.151 * MC1))) + 6.4 * MC1)

in which TMPPRM is the estimated observation time dry bulb temperature of the air in immediate fuel contact, in degrees Celsius.

Intermediate Calculations:

CHI = (344.0 − QIGN) / 10.0

in which, if (CHI**3.6 * PNORM3) is equal to or less than PNORM1, then P(I) and the IC are set to zero.

Probability of Ignition:

P(I) = (CHI**3.6 * PNORM3 − PNORM1) * 100.0/PNORM2

in which P(I) is limited to the range of values from 0 to 100.

P(F/I) is a function of the spread component for that fuel model normalized to the value the spread component would have under a specific set of severe burning conditions (slope class 1; 20-ft wind 20 mph; herbaceous vegetation cured; woody vegetation moisture content at the pregreen level; and the 1-, 10-, and 100-hour fuel moistures 3.0 percent.) This function was derived empirically by Main and others (1982).

Normalized Rate of Spread:

SCN = 100.0 * SC/SCM

in which SCM is, in the developers' best judgment, the SC for which all ignitions become reportable fires. An SCM was calculated for each fuel model and is included as a fuel model parameter.

Probability of a Reportable Fire:

P(F/I) = SCN**0.5

Ignition Probability:

IC = IRND(0.10 * P(I) * P(F/I))

in which the factor 0.10 is required to limit the range of IC to 0 to 100.

Human-Caused Fire Occurrence Index

MCOI = IRND(0.01 * MRISK * IC)

in which MRISK is the human-caused risk. See Deeming and others (1977) for details about its evaluation.

Lightning-Caused Fire Occurrence Index

For information about this model, see the publications by Fuquay and others (1979) and Fuquay (1980). The model assumes that a thunderstorm traversing an area forms a corridor aligned with the storm's track that receives both rain and lightning. Flanking the rain-lightning corridor on both sides are areas subjected to lightning only.

The width of the rain-lightning corridor affected (STMDIA), the total width of the lightning-only and lightning-rain corridors (TOTWID), and the discharge rate (CGRATE) for cloud-to-ground lightning are functions of the NFDRS lightning activity level (LAL):

NFDRS	CGRATE	STMDIA	TOTWID
LAL	*strikes/min*	*miles*	*miles*
1	0.0	0.0	0.0
2	12.5	3.0	7.0
3	25.0	4.0	8.0
4	50.0	5.0	9.0
5	100.0	7.0	11.0
6	(LRISK = 100, LOI = 100)		

Duration of Lightning at a Point Within the Affected Area:

LGTDUR = −86.83 + 153.41 * CGRATE**0.1437.

Fractions of the Area Occupied by the Lightning-Rain and Lightning-Only Corridors:

(Lightning-rain) FINSID = ((STMDIA * STMSPD * LGTDUR) + (0.7854 * STMDIA**2.0)) /((STMDIA * STMSPD * TOTWID) + (0.7854 * TOTWID**2.0))

(Lightning-only) FOTSID = (1.0 − FINSID)

Rain Duration at a Point Within the Lightning-Rain Corridor:

RAIDUR = STMDIA/STMSPD

in which STMSPD is the translational speed of the storm in miles per hour. For the NFDRS, a constant speed of 30 mph is used.

Moisture Content of the 1-Hour Fuels Within the Rain Area:

FMF = MC1 + ((76.0 + 2.7 * RAIDUR) − MC1) * (1.0 − EXP(−RAIDUR))

The ignition component within the area affected by rain is calculated exactly as the IC except that FMF is used

instead of MC1. The IC for the rain-affected corridor is denoted as ICR and is a significant deviation from the Fuquay model. As a simplification, it was decided to use the NFDRS IC function rather than the ignition probability function developed for the model. The difference was judged to be minor. Also, the LRSF was introduced to account for area-specific fuel conditions that affect the fire-starting efficiency of the lightning (Bradshaw and others 1983).

Area Weighted Ignition Component:

ICBAR = ((FINSID * ICR) + (FOTSID * IC))/100.0

Lightning-Risk:

LRISK = CGRATE * LRSF

in which

LRSF is the lightning risk scaling factor. See Deeming and others (1977) for a complete description.

LRISK is limited to a numerical range of 0-100.

Lightning-Caused Fire Occurrence Index Computation:

If it is not lightning, or if it is raining at the time of the afternoon weather observation at the fire-weather station, 25 percent of the previous day's LOI is used to account for carry-over fires.

LOI = IRND(0.25 * YLOI)

in which YLOI is the previous day's LOI;
otherwise

LOI = IRND(10.0 * (LRISK * ICBAR) + 0.25 * YLOI)

in which the multiplier 10.0 scales the LOI such that the expected number of lightning fires per million acres increases by 1.0 for every 10 points of LOI. The LOI is limited to a value range of 0 to 100.

If LAL 6 is observed or predicted, the lightning risk (LRISK) and lightning-caused fire occurrence index (LOI) are set to 100.

Fire Load Index

FLI = 0.71 * SQRT(BI**2.0 + (LOI + MCOI)**2.0)

in which the BI is limited to 100 as is the sum of LOI and MCOI.

APPENDIX—Parameters for fuel models

Fuel model	Ratios and Fuel loadings[1]						Depth (ft)[8]	MXD (pct)[9]	HD & HL (Btu/lb)[10]	SCM[11]	WNDFC[12]
	1-h[2]	10-h[3]	100-h[4]	1000-h[5]	Wood[6]	Herb[7]					
A Western grasses	3000	——	——	——	——	3000	0.80	15	8000	300	0.6
(annual)	0.20	——	——	——	——	0.30					
B California	700	109	30	——	1250	——	4.50	15	9500	58	0.5
chaparral	3.50	4.00	0.50	——	11.50	——					
C Pine-grass	2000	109	——	——	1500	2500	0.75	20	8000	32	0.4
savanna	0.40	1.00	——	——	0.50	0.80					
D Southern rough	1250	109	——	——	1500	1500	2.00	30	9000	25	0.4
	2.00	1.00	——	——	3.00	0.75					
E Hardwood litter	2000	109	30	——	1500	2000	0.40	25	8000	25	0.4
(winter)	1.50	0.50	0.25	——	0.50	0.50					
F Intermediate	700	109	30	——	1250	——	4.50	15	9500	24	0.5
brush	2.50	2.00	1.50	——	9.00	——					
G Short needle	2000	109	30	8	1500	2000	1.00	25	8000	30	0.4
(heavy dead)	2.50	2.00	5.00	12.0	0.50	0.50					
H Short needle	2000	109	30	8	1500	2000	0.30	20	8000	8	0.4
(normal dead)	1.50	1.00	2.00	2.00	0.50	0.50					
I Heavy slash	1500	109	30	8	——	——	2.00	25	8000	65	0.5
	12.00	12.00	10.00	12.00	——	——					
J Intermediate	1500	109	30	8	——	——	1.30	25	8000	44	0.5
slash	7.00	7.00	6.00	5.50	——	——					
K Light slash	1500	109	30	8	——	——	0.60	25	8000	23	0.5
	2.50	2.50	2.00	2.50	——	——					
L Western grasses	2000	——	——	——	——	2000	1.00	15	8000	178	0.6
(perennial)	0.25	——	——	——	——	0.50					
N Sawgrass	1600	109	——	——	1500	——	3.00	25	8700	167	0.6
	1.50	1.50	——	——	2.00	——					
O High pocosin	1500	109	30	8	1500	——	4.00	30	9000	99	0.5
	2.00	3.00	3.00	2.00	7.00	——					
P Southern pine	1750	109	30	——	1500	2000	0.40	30	8000	14	0.4
plantation	1.00	1.00	0.50	——	0.50	0.50					
Q Alaskan black	1500	109	30	8	1200	1500	3.00	25	8000	59	0.4
spruce	2.00	2.50	2.00	1.00	4.00	0.50					
R Hardwood litter	1500	109	30	——	1500	2000	0.25	25	8000	6	0.4
(summer)	0.50	0.50	0.50	——	0.50	0.50					
S Tundra	1500	109	30	8	1200	1500	0.40	25	8000	17	0.6
	0.50	0.50	0.50	0.50	0.50	0.50					
T Sagebrush-grass	2500	109	——	——	1500	2000	1.25	15	8000	73	0.6
	1.00	0.50	——	——	2.50	0.50					
U Western pines	1750	109	30	——	1500	2000	0.50	20	8000	16	0.4
	1.50	1.50	1.00	——	0.50	0.50					

[1]For each fuel model, the top value is surface-area-to-volume ratio (ft^{-1}), and the bottom value is fuel loading (tons/acre).
[2]1-hour timelag dead fuel moisture class.
[3]10-hour timelag dead fuel moisture class.
[4]100-hour timelag dead fuel moisture class.
[5]1000-hour timelag dead fuel moisture class.
[6]Live fine woody fuel class.
[7]Live fine herbaceous fuel class.
[8]Effective fuel bed depth.
[9]Assigned dead fuel moisture of extinction.
[10]Dead and live fuel heat of combustion.
[11]Assigned spread component value when all ignitions become reportable fires.
[12]Wind reduction factor from 20-foot standard height to the midflame height.

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Updating the National Fire-Danger Rating System (NFDRS) was completed in 1977, and operational use of it was begun the next year. The System provides a guide to wildfire control and suppression by its indexes that measure the relative potential of initiating fires. Such fires do not behave erratically—they spread without spotting through continuous ground fuels. Estimates of fire potential have a basis in the mathematical models used for fire behavior. The fire manager must select the fuel model that best represents the fuels in the protection area. Among the 20 fuel models available, not more than two or three are appropriate for any one area. This documentation of the 20 fuel models and their equations supplements previous reports on the System. The equations are presented in the coded format of FORTRAN and BASIC computer languages.

Retrieval Terms: fire modeling, fire occurrence, fire-danger indexes, forest-fire behavior, forest-fire risk, fuel moisture, fuel models.

United States
Department of
Agriculture

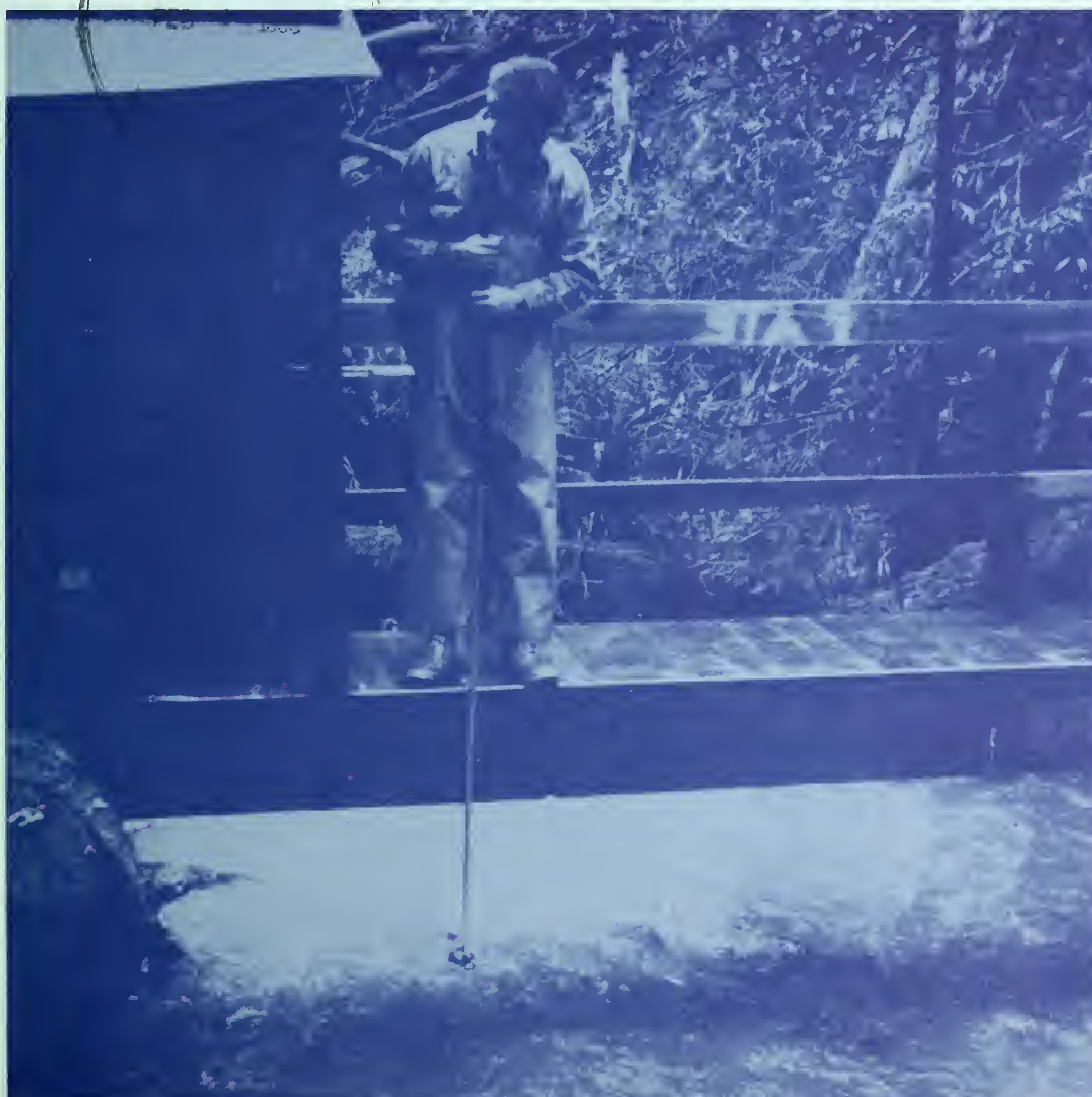
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-83

Measuring Suspended Sediment in Small Mountain Streams

Robert B. Thomas



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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

July 1985

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CONTENTS

Land management agencies are under increasing pressure to monitor the effects of their activities. One consequence has been a concern about the amount of sedimentation in streams near logging operations in forest stands. Sedimentation can adversely affect water quality as well as fish habitat. The Forest Service, U.S. Department of Agriculture has responded to this concern by increasing its collection of data on suspended sediment in streams. However, a unique set of problems exists when suspended sediment concentration is measured in forested catchments—especially those in which rain is the dominant form of precipitation.

Most suspended sediment moves during infrequent high flows that collectively account for only a small portion of the measurement period. The associated high transport rates and variances dictate that most data be collected during high flows, but the infrequency and brevity of the high flow periods combined with measurement and access problems cause acute problems in collecting data.

These problems will not be solved merely by increasing the amount of suspended sediment data collected. The collection process is both complex and expensive. Therefore, it is vital that such efforts yield maximum return. Often, data are collected without giving adequate thought to identifying the objectives or the administrative and technical problems that largely define what can be measured and how to measure. In such situations, analyses are difficult and interpretation ambiguous. No matter how sophisticated, analysis and interpretation can never substitute for well collected data.

This report describes the administrative and technical problems that define what to measure and how to measure suspended sediment in small mountain streams. It examines the factors that govern the quality of data collected in a monitoring program, with particular attention to use of automatic pumping samplers.

STATISTICAL CONSIDERATIONS

Hydrologists measure suspended sediment to compare one set of values with another. For example, they collect data above and below a logging site to determine how disturbance affects sediment concentration by comparing two sets of values. When checking for standards compliance, they compare their measurements to the standard. Even when "background" levels are being established, future data will be collected to compare with these levels.

A major concern is how such comparisons can be made "fairly." Hydrologists should be sure that any detected differences result from real differences between conditions and not from unrecognized factors in the measurement process. Keeping such factors to a minimum is not simple. Hydrologists must pay constant attention to details to ensure that they do not inadvertently introduce any unintended effects.

Data sets are formally compared by one of several standard statistical techniques. Such techniques must be used because the data are samples from larger populations that are impossible or impractical to measure totally. Each technique is based on a set of assumptions. Data collection must conform to these assumptions—at least approximately—for the test outcome to perform as designed. These assumptions bear not only on the nature of the measured phenomena, but also on how they are measured. The procedure used to choose values or items for inclusion in the sample is at least as important as the actual mechanics of measuring.

Measuring suspended sediment in streams is really sampling. Compared with continuous flow records that are essentially complete, only a very small proportion of the potential suspended sediment samples are actually collected and measured. Because inferences are based on samples, the procedure by which samples are selected greatly affects the properties of the inferences made.

Most data can be placed into one of two classes—time series or independent. The first type depends fundamentally on time or some other continuum along which the data occur. This class is characterized by dependence among data close together along the continuum, and by the fact that the order in which the data are sampled is of central importance in the analysis. The parent sequence and the sample taken from it are both called *time series*. Time series are usually sampled at equal time intervals, and the major sampling consideration is the length of the interval.

The second type of data is more common and does not depend on time. Instead, sample elements are chosen according to a formal random process that ensures statistical independence. Samples of this type are referred to as *independent*.

Elements of a population should not be selected purposefully for inclusion in a sample. Estimates of basic population characteristics based on purposeful samples are often "biased;" that is, the sample estimates of means and variances are *systematically* distorted from their population values. This means that the averages of the estimates of means or variances from all samples that could be collected do not equal their population values. This systematic distortion of estimates applies to the samples as a group rather than to any individual sample.

Most data sets of suspended sediment concentration in small mountain streams are not subject to time-series analyses because of the sporadic nature of the process and acute difficulties of measurement. This problem may be partially solved by the recent increase in the use of automatic pumping samplers that can be operated at equal intervals of time or flow. An excessive amount of data would be collected, however, if data are taken during low

flows with a time-based sampling interval short enough for describing variation during high flows. Therefore, time series analyses of sediment data are likely to be used only for special theoretical investigations, and independent samples used for application-oriented studies.

Traditionally, suspended sediment data have been collected opportunistically and treated as though they were independent data. Considering the large variation in such measurements and that samples were infrequent, perhaps the dependence structure of the sediment delivery process is not too critical for crude estimates. However, more samples need to be collected during high flows than is usually done.

One strategy widely used with independent data is stratification. The population is partitioned into several strata, each of which contains relatively homogeneous measurements of interest as compared with the total population. An efficient plan can then be developed to distribute sampling effort where most needed on the basis of stratum size and significance as well as variance. Prior knowledge about the structure of a population is better used in defining strata than in selecting a purposeful sample.

MEASURING SUSPENDED SEDIMENT

Suspended sediment is commonly measured with devices ranging from a vessel dunked in a stream to automatic pumping samplers. Regardless of the device used, the size of each water sample taken is extremely small. Consider taking one 500-ml water sample every hour from a stream flowing at 0.01 m^3/sec. For a sampling frequency and water sample size that are high by current standards, and for low discharge and suspended sediment concentration, this water sample represents only about 0.0014 percent of the flow! Another way to look at this is that of the 71,429 samples of 500 ml, that could have been collected during one hour's discharge, only one is selected. In sample survey work a 1 percent sample is considered small, but in this example under favorable conditions, the sample is only about 1/700th of that. For higher discharges—when suspended sediment concentration is higher and, therefore, more critical—or lower sampling frequencies, the proportion sampled is even smaller.

A small sample size is not necessarily inadequate; a small sample may be adequate if the variance of the process measured is small. But suspended sediment data sets from small mountain streams tend to have large variances, at least when considered across a wide range of flow conditions. It is this high variance, which increases with increased discharge, that causes most data collection problems. Suspended sediment concentration data from undisturbed watersheds in rain-dominated areas are characterized by periods between storms when concentration is relatively low and steady, interspersed with storm periods that produce wide fluctuations in discharge and concentration. In snow-dominated regions, these fluctuations are not as great, but the relationship between level and variation remains. Management activities can make this pattern more pronounced.

As an aid in planning suspended sediment measurement programs, consider the process of sediment transport and the associated physical quantities. At a stream cross section, a flux of both water and suspended sediment can be expressed as volume or mass rates for any instant in time. Water discharge is usually expressed as a volume rate and is represented as a time function on a hydrograph. An analogous time function gives the mass rate of sediment transport. The ratio of the suspended sediment mass rate to the water volume rate for a specific time is the mean suspended sediment concentration in the cross section. Three quantities of interest change over time—water discharge, suspended sediment transport, and suspended sediment concentration—each with its own actual or conceptual graphical representation.

For sediment studies, defining the suspended sediment transport process would be sufficient, but this process cannot be measured directly. A second approach is to approximate the transport process by multiplying water discharge and suspended sediment concentration measured continuously. It is not possible to measure instantaneous mean cross sectional suspended sediment concentration, and is difficult and expensive to measure continuous suspended sediment concentration at a point. The usual compromise is to measure suspended sediment concentration periodically at a point or along verticals in the cross section and use those values to estimate the mean concentration in the section as a whole.

In turbulent mountain streams, mixing can be fairly complete, so that suspended sediment concentration will be similar throughout a cross section except near the bed. In slower streams, however, the concentration at a given point in a section may not be the same as the concentration at another point or as the mean concentration for the whole section. Channel morphology, velocity gradients, and sediment size distribution combine to produce this variation. Each point in the cross section is associated with a time function, giving the concentration at that point which may be similar to but is not identical with the one for the stream as a whole. Each measurement, therefore, should either cover the section adequately to allow a good estimate of average concentration, or should come from a point with a known relationship to this average.

Also, cross sections at different locations on the same stream cannot be expected to have the same concentration time series even when close together. Downstream, more water and new sediment sources, combined with changes in channel morphology, can produce changes in patterns of sediment mobilization, suspension, and deposition.

These conceptual time function and associated graphical representations are useful tools for focusing the hydrologist's attention on data collection problems and on the disparity between what one intends to measure and what actually is measured.

When collecting suspended sediment data, the use of different sampling schemes or different measuring devices for the several conditions to be compared can bias measurements. By sampling various flow classes at different relative frequencies before and after a logging operation, for example, a false "difference" could be introduced solely or in part by the sampling system itself. The

different and complex hydraulic properties of measuring devices virtually ensure that comparing them will be subject to inherent differences and, therefore, will be suspect. Two primary ways to overcome this problem are either to use the same measuring devices and techniques at all stations, or to calibrate to a standard.

Where to Measure

The question "where to measure" usually has a simple answer because most often a particular activity dictates, in large degree, where data must be collected. Occasionally, however, the suspended sediment concentrations or loads of an entire area need to be characterized, so the more complex question arises of what watersheds to measure and where to measure them. The answer depends largely on the information wanted and the particular situation being studied. Often these kinds of investigations have poorly defined goals, so general prescriptions for sampling networks cannot be given. For such projects the hydrologist should carefully consider how the data will be used, and what questions they are intended to answer, and then get statistical help in designing an appropriate sampling plan.

When a specific disturbance is being monitored, the question of where to measure in the local sense is more pertinent. Generally, measurements should be made downstream as close to the disturbance as possible. The effect of a sediment-producing condition is attenuated and its effect confounded with the effects of dilution and other sediment sources farther downstream. If the downstream effects of a disturbance are being studied, it is best to measure at the site being affected.

Suspended sediment concentration data are often collected above and below a disturbance and the difference between stations taken as a measure of change caused by the activity. This can be a useful approach, but it is often done in a way that gives equivocal results. It may be assumed that suspended sediment concentration patterns at the two stations are identical, that "background" levels are uniform along the stream, and that any measured differences can be attributed to effects of the disturbance. This is not always a valid assumption, however, and should be tested in all cases by monitoring the two stations before the activity is begun. Usually, a regression relationship can be developed to estimate the "undisturbed conditions" response at the lower station given a measurement at the upper station. This relationship can be compared to actual levels measured at the lower station after the management activity has occurred. If the background levels at both stations are shown to be the same, comparisons can be made directly.

Such comparisons of "before" and "after" require more time and effort—something not often available before logging or some other change. The fact remains that if the disturbance is measured only after it occurred, any indicated differences between effects of the "treatment" and natural differences at the two stations cannot be apportioned. The activity may have caused the change, but there may have been a difference between stations before treatment—which posttreatment data cannot indicate. In some such cases, the best monitoring decision may be to forego measuring because collecting data that can give no valid results is a waste of money and effort.

Where to locate measurement stations also depends on the hydraulic conditions in streams. Often found in mountain streams are pool/riffle sequences (Leopold and others 1964). These generally stable features result from interactions between storage and transport of coarse sediment, and hydraulic control from channel bends, bedrock outcrops, and large organic debris. The conditions that exist at different positions within a pool/riffle element can be expected to affect not only the suspended sediment concentration, but flow measurements as well.

Paired stations should be set up in the same relative position within their respective pool/riffle elements. It is best to measure just above the downstream end of a pool where the stream spills over the crest into the riffle below. This crest acts as a control section having relatively uniform sediment and flow profiles thereby making it easier to collect accurate data. During high flows, however, control may be affected by bank conditions and channel morphology. Therefore several pool/riffle elements should be investigated to select one that will maintain control through an adequate range in flow. In some streams, control can be effected by geology or large organic debris. Bedrock cropping out at the crest of major riffles or falls can provide excellent control in natural channels. In some situations, well emplaced logs stabilize channels and provide good locations to measure both suspended sediment concentration and discharge.

If suspended sediment concentration characteristics vary up and down stream according to position within the pool/riffle sequence, care must be taken to occupy the exact same cross section for all measurements at a station. Small changes in position of the intake of the measuring device—either up and down stream, or within a cross section—can adversely affect measurements. Reliable markers should be installed at the stations so that data will be collected at the same places each time to remove this avoidable source of difference.

When to Measure

Once stations have been established, the data collection system must be decided on. Questions about sampling protocol are the most often asked and the most difficult to answer. A complete answer depends on the purpose for which data are taken, the analysis techniques to be used, and the characteristics of the particular set of data being collected. Short of defining the ideal sampling system, several general principles can help the hydrologist improve data collection. One general principle involves the relationship between flow and suspended sediment concentration.

Discharge and Suspended Sediment Concentration

In most streams suspended sediment concentration is strongly correlated with discharge. Large river systems usually contain abundant channel materials available for movement, so the energy of water discharge is often a good predictor of concentration. Streams draining small mountain catchments, however, often depend for their suspended load on episodic contributions

of fine materials from banks and upland areas. In these "event response" streams (Yaksich and Verhoff 1983) suspended sediment concentration depends on supply as well as discharge, and so they tend to have poorer relationships between suspended sediment concentration and flow. Nevertheless, discharge remains the best commonly measured correlate of concentration, and is useful as a guide to sampling.

The flow/suspended sediment concentration relationship is often exploited to estimate total suspended sediment yield for a period of record. Numerous discharge/concentration pairs are measured across a range of flows to form a "rating curve" that is usually a power function of the form:

$$C = aQ^b, \tag{1}$$

in which C is suspended sediment concentration, Q is discharge and a and b are parameters. This function can be used with the streamflow record or means of flow duration classes to estimate the total suspended sediment yield for a period.

One consequence of this relationship can be seen by selecting, for a period of record, a "large" number of discharge classes containing equal volumes of flow. The product of this volume and the mean sediment concentrations estimated from the rating curve using the midclass discharges estimates the total sediment yielded in each class. Because the water volume is constant across classes, the sediment volume increases for higher flow classes depending on the rating curve. When $b > 1$, which is the usual situation, this increase can be quite dramatic. It is not unusual to find situations where more than one-half of the sediment is carried by high flows that account for less than 15 percent of the water volume and that occur, perhaps, 2 percent of the time. Another factor characterizing suspended sediment concentration data in the high flow classes is their increased variability. Both of these factors indicate that most sampling should take place during high discharge.

Perhaps the most common shortcoming of existing suspended sediment data sets from small catchments, however, is the lack of measurements taken during high flows. Large storms increase the discomfort of taking measurements, make access difficult, and produce hazards because of high discharge. Also, the timing of high flows is often difficult to predict when people are not at the station. Nonetheless, data collected during high flows are essential to the development of good sediment rating curves. Because of the infrequency of high flows and difficult logistical problems, it is almost impossible to get adequate high flow measurements from hand sampling alone.

The advent of automatic pumping samplers and their increasing use encourage the hope that some problems of suspended sediment concentration data collection at high flows may be overcome. Unfortunately, pumping samplers have their own set of problems, some of which will be discussed in a following section. Still these samples afford the opportunity to collect data where little have been collected before, and to improve estimates of total suspended sediment loads. This additional information can also be expected to affect the design of sampling programs.

Sample Allocation

Data collection efforts must be distributed throughout flow classes in such a way that classes with (a) a large volume of sediment, and (b) high variance are most frequently sampled. Sampling these classes is similar to the allocation problem in stratified random sampling with finite populations if the flow classes are thought of as strata.

The basic (Neyman) allocation formula (Cochran 1963) is:

$$P_i = \frac{N_i\sigma_i}{\sum_{j=1}^{k} N_j\sigma_j} \tag{2}$$

in which P_i is the *proportion* of the sample taken in the i^{th} stratum, $N_i(N_j)$ is the size (that is, number of objects) and $\sigma_i(\sigma_j)$is the standard deviation in stratum i(j), and k is the number of strata. For each stratum the product of its size and its standard deviation is formed; the proportion of the sample to take in that class is this product divided by the sum of the corresponding products across all strata.

The Neyman allocation was applied to sampling large "event response" rivers where daily suspended sediment yields can be characterized by a single sample (Yaksich and Verhoff 1983). For their sampling, N_i is the number of days in the i^{th} flow class. For small mountain catchments, however, variation in suspended sediment concentration is too large to use daily values and shorter time periods are not practical without automatic sampler control equipment. Another technique therefore must be used. Because the purpose is to estimate the quantity of suspended sediment in each flow class, it is reasonable to use suspended sediment yield in class i as a measure of size in place of N_i. The analogy is not strictly correct, but the approach will give a useful allocation.

If pertinent suspended sediment concentration data are available, sample estimates of σ_i will allow equation 2 to be used directly. More often such values are not available. One surrogate for the standard deviation is the *range* of values in a class—either suspended sediment concentration or discharge at flow class boundaries (Murthy 1967). Flow and sediment volume curves, however, often result from several sediment rating curves developed for different time periods or hydraulic conditions. Therefore, there may be no unique suspended sediment concentration associated with a flow volume class boundary. Because discharge is readily available, it can be used instead. Using the range of discharge to estimate the standard deviation deemphasizes the proportion of samples to be allocated to the higher flow classes.

Suppose S_i is a measure (mass or percent) of the sediment in class i, and R_i is the range of discharge in the i^{th} class, the allocation formula becomes approximately:

$$P_i \cong \frac{S_iR_i}{\sum_{j=1}^{k} S_jR_j} \tag{3}$$

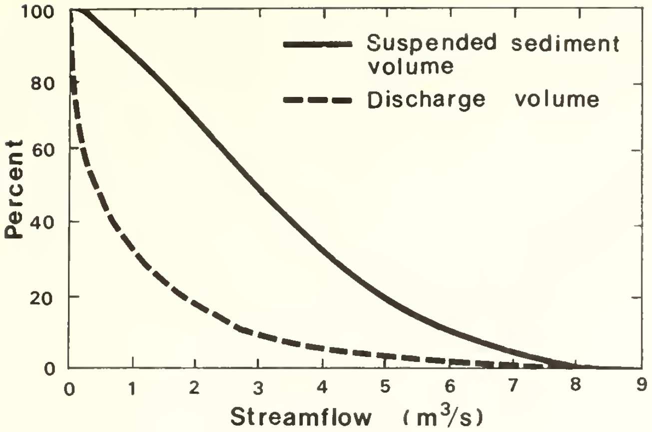

Figure 1—Percentage of suspended sediment and discharge volumes occurring at greater than indicated streamflows, Caspar Creek, northern California (Rice and others 1979).

Using equation 3 is still difficult because quantities to be estimated from the sample are required to estimate sample size. Prior data on the same or similar streams may exist that can be used to form a suspended sediment volume curve from which estimates of the volume of suspended sediment coming from each flow class can be obtained. It is sometimes necessary, however, to set up an arbitrary sampling plan for a year or so to get preliminary information that can be used to modify the plan later.

The application of equation 3, can be illustrated by using data collected from Caspar Creek in northern California (Rice and others 1979) (*fig. 1*). The percent axis was partitioned into 20 classes each containing 5 percent of the flow volume. Discharge rates corresponding to class boundaries were then read from the graph and percentages of sediment delivered at flows greater than these values were determined. (Percentages or proportions of sediment can be used in place of actual volumes or weights as long as they are used consistently.) For this stream the allocation indicates that samples should be heavily concentrated in the higher flow classes; in fact, it shows that nearly three quarters of the measurements should be taken in the highest flow class alone. This is not surprising when it is considered that more than half of the range of flows occurred here as well as 30 percent of the suspended sediment volume, so this class contained a large portion of the suspended sediment as well as being highly variable.

Each class should be allocated a minimum of about five samples so that mean discharge for the class and its variance can be estimated. For example, if 100 samples are to be collected, most classes in *table 1* are too small. By experimenting with the number of discharge classes and their boundaries an allocation can be produced that has at least five samples in each class. To accomplish this allocation, the two highest flow classes were left intact and the next two, the following three, and the last 13 classes were grouped to form a five-class allocation *(table 2)*. With 100 samples under this allocation, six would go into each of the two lowest flow classes.

A minimum number of samples in the low flow classes was obtained at the expense of samples previously allocated to the high flow classes. This is because the ranges and sediment contributions of the composite classes increase, making the denominator of equation 3 larger. A practical compromise must be

Table 1—*Data and calculations to establish suspended sediment sample allocation using data from Caspar Creek (Rice and others 1979)*

Water volume yielded at greater than indicated flows (pct)	Flows at class boundaries (m^3/s)	Sediment volume yielded at greater than indicated flows (pct)	R_i Flow range (m^3/s)	S_i Sediment volume in class (pct)	R_iS_i	P_i (pct)
0	8.5	0				
5	4.1	30	4.4	30	132.0	73
10	2.7	55	1.4	25	35.0	20
15	2.1	64	0.6	9	3.6	2
20	1.7	72	0.4	8	3.2	2
25	1.4	79	0.3	7	2.1	1
30	1.1	84	0.3	5	1.5	1
35	0.9	87	0.2	3	0.6	
40	0.7	91	0.2	4	0.8	
45	0.6	93	0.1	2	0.2	
50	0.4	95	0.2	2	0.4	
55	0.3	96	0.1	1	0.1	
60	0.2	97	0.1	1	0.1	
65	0.2	97	<0.1		<0.1	1
70	0.1	98	<0.1		<0.1	
75	0.1	98	<0.1		<0.1	
80	<0.1	99	<0.1		<0.1	
85	<0.1	≅100	<0.1		<0.1	
90	<0.1	≅100	<0.1		<0.1	
95	<0.1	≅100	<0.1		<0.1	
100	0.0	100	<0.1		<0.1	
					$\sum R_iS_i$ = 179.8	

Table 2—*Calculations for five class suspended sediment sample reallocation formed by grouping classes in table 1*

Water volume yielded at greater than indicated flows (pct)	Flows at class boundaries (m^3/s)	Sediment volume yielded at greater than indicated flows (pct)	R_i Flow range (m^3/s)	S_i Sediment volume in class (pct)	R_iS_i	P_i (pct)
0	8.5	0				
5	4.1	30	4.4	30	132.0	63
10	2.7	55	1.4	25	35.0	17
20	1.7	72	1.0	17	17.0	8
35	0.9	87	0.8	15	12.0	6
100	0.0	100	0.9	13	12.0	6
					$\sum R_iS_i$ = 208.0	

struck between having many classes to reduce the overall variance and having fewer classes so that each class will contain at least a minimum number of samples. For most suspended sediment sampling programs five or six strata should be adequate.

The primary benefit of using equation 3 is to emphasize the need to collect suspended sediment concentration data at higher flows and to estimate *approximate* proportions of sampling effort to expend at different flows. This technique should be kept in perspective and not used slavishly. Scenarios based on synthesized flow and sediment volume curves can be developed that bracket a particular set of field conditions. Applying the equation to these scenarios can help develop reasonable sampling programs that should serve to emphasize the need to sample more heavily in the higher flow classes than is typically done.

Several factors are not addressed by equation 3. One of these is the total sample size. Formulas to estimate total sample size in stratified random sampling are available (Cochran 1963), but either cost information or specification of the variance of the total suspended sediment volume are required, in addition to variance and size estimates in all strata. This complex of assumptions strains credibility in applying the total sample size formulas to this situation. In any event, sample size usually depends on constraints on time, funds, and personnel.

Equation 3 does not indicate when to take the prescribed number of samples in each class. For the mean and variance estimating formulas to be strictly correct, the data in each class should be randomly selected. As a practical matter, however, such selection is seldom possible, and requires automatic sampling control equipment that is not generally available. As an accommodation until such control techniques are accessible, samples should be distributed more or less evenly throughout the time period in each class rather than being clustered during a few intensive episodes. This requirement is particularly important for high flows that should be sampled more frequently, but that occur infrequently.

Sampling at regular intervals in high flow classes may help reduce the effect of dependence as long as the sampling interval is not too small. It is in these classes where automatic equipment is most useful because the logistics of measuring high flows by hand in most studies of small mountain streams is virtually insurmountable. Some provision is required for sampling less frequently during lower flows to avoid collecting excessive data where they are not needed. Making the automatic sampling interval dependent on stage is one solution to this problem (Hicks and Bright 1981). Collecting samples at equal intervals of water discharge (or estimated sediment discharge) ensures that data will be better distributed throughout flow classes, but is more difficult to accomplish.

Sources of Variation

Suspended sediment concentration values vary because of several factors. Some of these are at least partially predictable, but others are not. The unpredictable factors are collectively called *random variations*, and these are large in most suspended sediment sampling programs. The effects of random variation on the estimates of means and variances are usually reduced by collecting a larger sample. This is one factor that should increase allocation of sampling effort to high variance strata.

Partially predictable factors are sometimes ignored and remain to add to the random errors, but they can be used to partition a data set into subsets, each of which represents the process under particular conditions. For example, separate rating curves can be developed for rising and falling flows. This approach is based on recognition that rising and falling flows of the same magnitude often show quite different suspended sediment concentration (Walling 1977). Plotting suspended sediment concentration/discharge pairs for a storm period often shows a classical "hysteresis" effect in which the trace advances along one path, but returns to prestorm conditions along another (Gregory and Walling 1973). This condition implies a fundamental if temporary change in the deterministic component of the rating curve.

Rating curves also undergo long-term changes. A commonly observed change over a season is that suspended sediment concentration at a given stage will usually be higher early in the runoff season than later on (Gregory and Walling 1973). Also, major events can permanently change the response of concentration at a given station. The hydrologist will have to decide for each situation how to partition a data set to best represent the particular conditions but must keep in mind that an improvement

in sensitivity will be at the expense of greater analytical and sampling efforts.

The previous discussion tacitly assumed that suspended sediment concentration data collected in the usual way are independent rather than time series data, which is actually the case. If the measured values are not too close together in time, their dependence may not severely distort the estimates. It will eventually be practically feasible to measure concentration at regular intervals of flow, to measure concentration continuously, or to employ techniques to obtain independent samples from time series processes. Until then the best strategy is to distribute the data collected within each flow class as evenly as possible.

Another point concerning time of measurement refers to the "above and below" monitoring situation introduced earlier. In addition to ensuring that the stations are located at similar points in their respective pool/riffle sequences, and that the stations are calibrated before disturbance, accounting for the problem of variation over time is needed. Why suspended sediment concentration varies quickly over time in small streams is not well understood. Assigning a cause for a particular spike in concentration is often difficult.

The intent to equalize measurement conditions suggests that both stations should be measured simultaneously. An objection could be made that whatever sediment is measured at the upper station would take time to arrive at the lower station requiring a lag time between stations proportional to average velocity. Basing sampling at the lower station on the time it takes a water mass to travel from the upper station, however, raises practical and theoretical problems. Velocity and, therefore, lag time, varies with discharge and there is no simple way to measure average velocity between stations to decide when to operate the lower sampler. Because both sediment and water can be added and sediment can be removed from the flow in a reach, it is not even clear what is meant by measuring the same water mass at the two stations. It is probably best to calibrate by measuring at both stations simultaneously or at a constant time difference—to allow the data collector to travel from the upper to the lower station—and relying on enough measurements throughout each flow class to give valid comparisons. By matching measurements in this way, the variance is reduced and more sensitive comparisons can be made with fewer observations.

A final aspect concerns how long monitoring should be carried out both before and after a treatment. This depends on several factors, chief among which are the purposes of the study, the characteristic changes in the suspended sediment regime produced by a treatment, and the pattern of storm impacts. Each application presents unique problems so it is difficult to generalize, but several important conditions can be mentioned.

Perhaps the most stringent requirement is that the pretreatment and posttreatment monitoring include a wide and similar range of flow conditions. Otherwise only part of the suspended sediment concentration/discharge relationship will be measured for prediction or comparison. This means that these periods need to be long enough to allow a reasonable chance that the vagaries of climatic variation will produce storms of the required size. This will vary by locality and should be estimated for each individual case.

Also, the characteristics of a particular land treatment applied at a particular site can affect the posttreatment monitoring program. The sources of suspended sediment production may develop immediately and heal over time such as loose sediment being directly pushed into the channel system by roadbuilding. Or, the sources may be delayed as would be the case where debris is produced from logging-caused landslides which occur only after decay of the root matrix. Even if these changes are anticipated there is no assurance that storm inputs of sufficient range in magnitude will occur during the period of vulnerability or measurement.

All of these factors make the outlook for planning a successful suspended sediment measurement program problematical. The primary lessons are that the establishment of such a program is not a simple undertaking. If conditions and available monitoring time are such as to preclude a reasonable chance for collecting adequate data, the measurements would best not be made.

Measuring at a Cross Section

Data Collection

To estimate total suspended load from occasional measurements of suspended sediment concentration, water discharge information is necessary. Discharge must be known when suspended sediment concentration is measured to develop suspended sediment rating curves, and continuous discharge is required when suspended sediment yield is estimated.

A related variable sometimes measured is turbidity. In some situations it is recorded for its own sake; usually because standards restrict increases in turbidity. Another use for turbidity data is to estimate suspended sediment concentration by relating the two quantities and using the more easily measured turbidity to estimate suspended sediment concentration at times when it has not been measured directly (Beschta 1980, Walling 1977). The relationship between these two quantities is not perfect, nor should turbidimeters be calibrated in units of suspended sediment concentration. For many streams and for a wide range of particle sizes, however, a close statistical relationship can be developed, often one that is far better than the relationship between suspended sediment concentration and discharge. The relationship tends to be poorer for large particle sizes. The suspended sediment concentration/turbidity relationship must be developed separately for each measurement station by collecting a set of pairs of measurements of both quantities over a wide range of flows. It should also be checked periodically to detect changes in the relationship over time. The suspended sediment concentration values can then be regressed on the turbidity data to obtain a prediction equation. Even though this technique has problems, the opportunity for improving estimates of suspended sediment totals and for better defining suspended sediment concentration variation (especially with the use of continuously recording turbidimeters) justifies more frequent field application.

The great difference in suspended sediment concentration that can exist at different points throughout the cross section should also be considered. To collect a sample that has concentration similar to that existing throughout the stream cross section, a

sampler is needed that integrates partial samples from many points in the section; that is, adequate areal *coverage* of the cross section is needed to ensure that a sample with concentration and particle size properties similar to those in the stream is collected.

A set of standard suspended sediment measuring devices has been developed by the Federal Inter-Agency Sedimentation Project (FIASP 1963) along with procedures for their correct use (Guy and Norman 1970). Among these devices, the DH-48 depth-integrating sampler is probably the most widely used by forest hydrologists for sampling small streams. It can be used from a low bridge or when wading. With its nozzle pointing upstream, the sampler is lowered at a uniform rate into the flow from the surface to near the bottom and returned to the surface for each of a set of verticals across the width of a stream. Air is exhausted from the sample collection bottle to admit the water/sediment mixture at stream velocity. This condition of identical stream and nozzle velocities—isokinesis—ensures that the sample is the same concentration as that in the stream near the nozzle and that all conditions are proportionally represented. Careful use of the DH-48 sampler with enough verticals will ensure a sample with a concentration approximating that in the stream cross section. The DH-48 (or similar device designed by FIASP) will be considered in this report as the standard instrument with which to compare other suspended sediment concentration measuring devices.

Used properly, the DH-48 sampler provides a satisfactory measure of the suspended sediment concentration in a cross section, but wide flow coverage is difficult because people must be present to make measurements. Conversely, automatic pumping samplers, which are becoming increasingly available, can be operated in either a time- or flow-dependent mode, thereby giving reasonable coverage of flow levels, but they sample only one point in a stream cross section. Although automatic samplers offer the prospect of more timely data, problems are associated with their use.

Pumping Samplers

A major problem with the pumping sampler concerns intakes: their type, orientation, and placement within a cross section. Some samplers have cylindrical intakes with a series of holes around their periphery and internal weights to help hold them in place on the streambed. These intakes are designed for wastewater sampling and are not well suited for suspended sediment measurement. Because sediment suspension depends heavily on velocity and because the hydraulics of a sediment suspension moving through one of these intakes is complex, it is difficult to quantify their effect on the sample. A better device is a nozzle similar to that on the DH-48. A nozzle can be made from a piece of thin-walled noncorrosive tubing with an inside diameter equal to that in the sampling hose, having an orifice beveled on the outside.

Hand samplers are oriented with their nozzles pointing directly into the flow so that the pressure of the incoming water will produce isokinetic sampling. Because pumping sampler nozzles remain in place for long periods of time between samples, pointing the nozzles upstream increases the chances of clogging. For accurate measurement of ambient suspended sediment concentration the nozzle should be pointed away from the flow rather than at any other angle except directly facing it (Winterstein and Stefan 1983). This direction gives about 90 percent or more of the "true" concentration across several particle sizes while greatly reducing the opportunities for clogging.

Placement of the pumping sampler nozzle in the stream cross section is also important because concentration gradients occur across and especially up and down the section. The ideal spot to locate the intake nozzle depends on stage, concentration, particle size distribution, and cross sectional configuration. The nozzle usually cannot be positioned in one place so that it will sample correctly for all conditions. The intake should be near the center of the stream, perhaps above the thalweg, and at least 10 cm or so above the bed to avoid sampling the saltating portion of the bed load. If the nozzle is a fixed distance above the bed, it will sample a different relative position in the vertical concentration gradient for different stages, thus changing the relationship between the sampled and actual concentration under varying conditions. A better approach is a device to sample at the same proportion of depth regardless of stage (Eads and Thomas 1983).

Each automatic sampler should be calibrated against measurements made with a DH-48. This requires visiting the installation over a wide range of discharge conditions to collect simultaneous measurements with a DH-48 and the automatic sampler. These pairs of values can be plotted to indicate how well the pumping sampler measures what a hand sampler would have measured. For a given installation, the DH-48 values can be regressed on those for the pumping sampler. The resultant equation, if it gives a satisfactory prediction, can be used to estimate suspended sediment concentration in the cross section from the automatic sampler data. Each automatic sampler installation measures something different—from other installations and from measurements made with depth integrated samplers. For correct comparisons of mean suspended sediment concentration between stations or for estimates of total suspended sediment transported through a cross section, pumping sampler values should be related to a standard.

Choosing and setting pumping sampler hose velocities are also problems. Some machines have no simple means for adjusting hose velocity. Even when the velocity can be regulated, stream velocities are constantly changing and no mechanism exists to alter the sampler accordingly. This means that if a pumping sampler nozzle is pointed upstream, isokinesis is possible only for the one preset velocity. For other nozzle orientations isokinesis is not possible so pumping velocities will have to be selected empirically and estimates based on calibration. The need to sample at high flows suggests setting hose velocities to sample correctly at stream velocities occurring at the intake during these flows. Although this practice will starve samples taken at lower velocities (Federal Inter-Agency Sedimentation Project 1941) they are not as critical and calibration will help improve these estimates. Also, from the standpoint of preventing sediment accumulation in the internal plumbing of the automatic samplers, it is better to err on the side of higher velocities.

Automatic samplers themselves cause problems that affect the data being collected. One such problem is cross-contamination; for example, a sample bottle may receive some of the sediment

that belongs to the previous sample and not get all of the sediment properly belonging to it. For a series of observations when concentrations are not changing rapidly, compensation occurs reducing the effect of cross-contamination. With wide swings in concentration, however, the true change can be greatly deemphasized and sample variances falsely reduced. This problem results primarily from the design of the particular sampler and is not directly under the hydrologist's control, except when a sampler is purchased (Thomas and Eads 1983).

PLANNING STUDIES OF SUSPENDED SEDIMENT

Planning is the key to successfully studying any natural phenomenon. Objectives must be set, intended analyses identified, and sampling schemes established and balanced against available resources. The data are then collected and analyzed, and interpretations made.

Studies of suspended sediment should also be subject to these same procedures, but they often are not. This is perhaps the most common reason why such studies fail. The long-term nature of many suspended sediment measurement programs, a general feeling that "monitoring" is not a formal study, and the lack of widely accepted analysis procedures may all contribute to the situation.

There are several characteristics of suspended sediment studies that should be particularly emphasized. Setting general and then more specific objectives is absolutely essential. There is no other way to select the analyses required and the consequent sampling program needed to collect adequate information. This must be done, in effect, when statistical analyses are performed (analyses answer only specific questions) and it is much better to decide before the data are collected when sampling procedures can be influenced. If definite goals are not clear at the outset, it may be because the study will not provide useful information. Suspended sediment "monitoring" should only be done when there are clear reasons for doing it.

Obtaining timely measurements of high quality represents a major portion of the effort going into a suspended sediment measurement program. The procedures and requirements must be defined in detail so that persons not present at the outset of a study will be able to collect comparable data later on. The planning should identify required instrumentation, techniques of use, and sampling protocol.

Many suspended sediment studies extend over long time periods. This fact increases the opportunity for personnel changes and for losing the intent of the experiment. It is essential, therefore, to document the study planning so that it can be reviewed periodically by those who set up the study and by those who may follow. The plan should be a working document wherein all of the history, reasons, and objectives for undertaking the study can be set down, the sampling plan described, and instructions given on measurement procedures, station location, and analyses to be performed. The hydrologist who prepares a complete and well thought out plan at the start of a program to measure suspended sediment goes a long way towards ensuring its success.

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The Forest Service, U.S. Department of Agriculture, is responsible for Federal leadership in forestry. It carries out this role through four main activities:

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The Pacific Southwest Forest and Range Experiment Station

- Represents the research branch of the Forest Service in California, Hawaii, and the western Pacific.

Thomas, Robert B. **Measuring suspended sediment in small mountain streams.** Gen. Tech. Rep. PSW-83. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1985. 9 p.

Measuring suspended sediment concentration in streams provides a way of monitoring the effects of forest management activities on water quality. Collecting data on suspended sediment is an act of sampling. The nature of the delivery process and the circumstances under which data are collected combine to produce highly variable results that are difficult to analyze and interpret. Data-collection stations are set up to compare one set of measurements to another. They should be located with regard to channel morphology. Deciding when to measure suspended sediment is a major problem in carrying out most studies. Concentration depends heavily on streamflow discharge, which can be used to allocate sampling resources to appropriate flow levels. Restrictions in budgets and technical concerns have fostered the increased use of automatic pumping samplers in measuring suspended sediment.

Retrieval Terms: suspended sediment, sampling, measurement, pumping samplers

United States
Department of
Agriculture

Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-84

Meadows in the Sierra Nevada of California: state of knowledge

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Acknowledgments:

I thank the following persons for their interest and help in defining goals and objectives for management of meadow resources, or for their review of the manuscript, or both: Leonard Topping, Madera County; John T. Stanley, Harvey & Stanley Associates Inc.; Jerry L. Neal, Robert Hall, and Rex Quinn, Sierra Club; Lewis E. Carpenter, Henry A. Doddridge, Ray Marxmiller, and Claude L. Brown, Fresno County Sportsmen's Club; Gordon C. Ashcraft and Ronald C. Bertram, California Department of Fish and Game; David J. Parsons and Steven H. DeBenedetti, Sequoia and Kings Canyon National Parks; Richard Riegelhuth, Yosemite National Park; Gordon C. Heebner and James R. Shevock, Sequoia National Forest; and John (Ken) Stithem and Jerome DeGraff, Sierra National Forest. Also, I thank Stanley E. Westfall who since 1965 has been involved in meadow research.

Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

September 1985

Meadows in the Sierra Nevada of California: state of knowledge

Raymond D. Ratliff

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IN BRIEF . . .

Ratliff, Raymond D. **Meadows in the Sierra Nevada of California: state of knowledge.** Gen. Tech. Rep. PSW-84. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, U.S. Department of Agriculture; 1985. 52 p.

Retrieval Terms: meadow classification, meadow productivity, meadow management problems, meadow conditions and trends, mountain meadows, Sierra Nevada, California

Management of mountain meadows in the Sierra Nevada of California to maintain or restore geologic and biologic stability, while providing amenities, is a common goal of managers and users. Meadows are wetlands or semiwetlands supporting a cover of emergent hydrophytes and mesophytes and dry herblands of the subalpine and alpine zones. These meadows are concentrated use points which, once destroyed, are not quickly replaced.

American Indians had little adverse effect on meadows of the Sierra Nevada. Moderate to light grazing by native animals was probably common. Ranching—the first industry in California—expanded as the Spanish missions became established. Cattle were valued for their hides and tallow. Sheep numbers remained low during the Spanish and Mexican periods. Then, the gold rush of 1849 ushered in a shift toward beef production, and large numbers of sheep as well as cattle were used for meat. As mining decreased, sheep ranching for wool increased; by the 1870's, California led the nation in wool production.

Summer grazing in the mountains began during droughts in the 1860's and 1870's. Sheep grazing soon became the dominant use of the meadows. Today, however, cattle have all but replaced sheep in the National Forests.

Overgrazing of meadows in the late 1800's and early 1900's resulted in widespread deterioration. Early attempts by the National Parks and Forest Reserves to regulate grazing were mostly fruitless. Effective control in the Forest Reserves began when grazing permits were required by the Forest Service, U.S. Department of Agriculture. Meadows have improved, but local abuse can still be found.

Recent concern over the conditions of meadows has resulted in studies by the Forest Service, U.S. Department of Agriculture; the National Park Service, U.S. Department of the Interior; and colleges and universities. Although much is yet to be learned, a body of technical knowledge focused on meadows of the Sierra Nevada is now available, serving as a source for guides to meadow management.

The basic unit for meadow classification is meadow site—an area of homogeneous species composition having a general species composition different from that of adjacent areas. Meadow sites are grouped by physiographic properties; hydrologic or major floristic properties, or both; and close floristic similarity into meadow subformations, meadow series, and meadow site associations.

Conditions favorable to accumulating fine-textured materials and establishing a shallow water table are prerequisite to meadow development. A characteristic condition is a favorable drainage area-to-slope relationship: large drainage areas with steep stream gradients do not contain meadows.

True pedogenic soil horizons are rarely found in soils of Sierra Nevadan meadows. Four main depositional units are present: pre-Holocene alluvium; a buried soil; stratified sand; and interstratified layers of grus, peat, and sandy loam. Layers of known age in the fourth unit indicate a soil accumulation rate of about 1.8 inches (4.7 cm) per century.

Herbage produced on Sierra Nevadan meadows in California ranges from less than 300 lb per acre (336 kg/ha) to over 4,000 lb per acre (4,484 kg/ha). High-elevation meadows in poor condition generally produce less herbage than low-elevation meadows in better condition. Mesic meadows are usually more productive—although production varies more from year-to-year—than meadows at the extremes of the moisture gradient.

Meadows respond to fertilization. Nitrogen can increase yields dramatically. Phosphorus, however, has been more consistent in increasing yields. Usually, nitrogen favors grasses and grasslikes and phosphorus favors legumes.

Poor sites respond proportionally more to fertilization than good sites.

Carbohydrate storage cycles of meadow species appear to follow the natural pattern: gradual decline in winter, sharp decline at the start of spring growth, and buildup during maturation. Defoliation in early spring or at flowering is detrimental to many meadow species. Late defoliation, preventing carbohydrate accumulation in developing shoots, may damage some species.

The need to leave some herbage ungrazed has long been recognized. For meadow sites in the Sierra Nevada, the rule of "graze half, leave half" is unsafe. Use of very wet or dry meadow sites should not exceed 35 percent; 45 percent use of more mesic sites is satisfactory.

Some of the more common conditions that can adversely affect meadows are:

- Defoliation of meadow plants. If too severe, too frequent, and at the wrong time, defoliation can deteriorate meadows. When properly coordinated with the requirements for growth and reproduction of a given species, grazing does not usually harm the ability of that species to produce herbage and survive.
- Preferential grazing. This major cause of range deterioration is caused when animals and humans use specific areas about the same time each season, and a treatment that favors one species may eliminate another from the stand. Breaking the use pattern by modifying frequency and timing of use is the only effective way to counter the harmful effects of preferential grazing.
- Trampling. This condition compacts the soil and cuts the sod. Even when the sod is not cut, trampling may lower the pH of the soil solution. Damage occurs when the stress applied exceeds the resistance of the soil to deformation. The most obvious signs of trampling damage are holes punched in meadows and multiple trails.
- Redistribution of nutrients. Nutrients become concentrated in certain areas. Livestock redistribute nutrients within and among meadows and closely associated ecosystems. People redistribute nutrients from distant ecosystems to meadows. Short-term effects of nutrient redistribution are evident as dung pats and urine spots. Long-term effects gradually become evident as changes in species composition.
- Rodent activities. Although rodent activities can markedly affect species composition and may induce erosion by channeling water, cultivation of the soil by rodents is beneficial. Overgrazing emphasizes the negative effects of rodents, but rodents inflict little harm to meadows in good condition.
- Invasion by lodgepole pine. This autogenic process is both aided and hindered by livestock disturbance. A mineral seedbed in a well-lit, warm, moist environment favors lodgepole pine seed germination. Early snowmelt and long snow-free periods favor seedling establishment. Deep, long-lasting snowpacks favor young trees.
- Fire. A critical element of meadow history and maintenance is fire. Major fires in their watersheds affect Sierra Nevadan meadows. Sediment depositions from major fires are evidenced by charcoal in many meadow soils. Light fires, which burn only the current growth, do little harm to trees or meadows. Hot fires, which burn mulch and peat, may kill well-established trees and greatly damage meadows.
- Erosion. This condition occurs naturally and as a result of overgrazing. Maintaining or restoring its hydrologic characteristics is essential to maintaining or restoring a meadow. Erosion control in meadows is designed to check gully progress, refill gullies, and restore the water tables. Good erosion control includes assessing meadow stability and causes of erosion, grazing management, building properly designed structures, planting appropriate vegetation, and controlling rodents.

Vegetal cover, including litter, is the characteristic most used to indicate soil condition. Species composition and ecological position are preferred to indicate vegetative condition. Meadow sites in excellent condition have a dense, even herbaceous cover (about 100 percent) and are composed mostly of decreaser species.

How a meadow site measures up is indicated by its condition. How management measures up is indicated by trend in condition. Management of meadows must be based on trust, agreement, and commitment of managers and users. The products to be produced from a given area and the actions needed to produce them must be decided. Good range management must be practiced and must include proper use, restoration efforts, monitoring condition trend, and user education.

INTRODUCTION

Meadows of the Sierra Nevada of California provide the bulk of forage on many grazing allotments, park preserves, and wilderness areas. And yet they comprise only 10 percent of the land area there. An abundance of rodents, insects, and reptiles provides food for many species. Ecotones—timbered edges—between meadows and forests contain many other animal species. Mountain meadows, therefore, are valuable for the production of livestock products, the maintenance of wildlife populations, and the grazing of recreation stock. Furthermore, they provide scenic vistas, and their timbered edges and grass carpets are favored campsites of forest, park, and wilderness visitors. Meadows filter sediment from water flowing from surrounding slopes, and in so doing provide clean water for human use and for maintaining suitable fish habitat in streams and lakes.

The diversity and richness of habitats available in meadows and their associated vegetations draws people as well as animals. Repeated high or untimely concentrations of use can cause damage to the resource. Interest in maintaining existing meadows, restoring deteriorated meadows, and managing all meadows properly is increasing. This interest is expressed in movements for more wilderness areas, and in concerns about wildlife, fish, and the quality of life. All these developments have led to the need to know more about mountain meadows.

This state-of-knowledge report summarizes available information about the meadows of the Sierra Nevada of California—how meadows are classified, the development of meadow soils, the productivity of meadows, problems in management, and ways to evaluate range conditions and trends.

BRIEF HISTORY

Livestock grazing and meadows of the Sierra Nevada have a common history of more than 100 years. And that history is well documented (Burcham 1957, Vankat 1970). Before the arrival of Spanish and Mexican colonists, only native grazing animals were present in California. Deer, elk, pronghorn antelope, and bighorn sheep were the only large ungulates. Except locally, deer, elk and antelope were not abundant in the Sierra Nevada. Meadows of the Sierra Nevada, therefore, probably existed under a regime of moderate or light grazing—mainly by deer, bighorn sheep, and small mammals.

American Indians of the area were hunters and gatherers. Their main influence on the native vegetation was through use of fire as an aid in hunting game. They did not much influence meadow vegetation. Use of fire by American Indians may, in some instances, however, have served to keep out invading trees and, thereby, an open meadow condition was maintained.

Grazing by Livestock

Cattle ranching was the first industry in California. It started in 1769, at San Diego, with about 200 head. As missions were established, the cattle industry expanded. At their zenith, the missions each had several ranches—some for cattle and others for sheep and other livestock, and claimed about one-sixth of present-day California. In addition to mission livestock, individually owned herds of livestock on ranches and great numbers of feral stock grazed the land.

Cattle were valued chiefly for their hides and tallow during the Spanish and Mexican periods (1769 to 1850). Emphasis toward meat production changed markedly as an outgrowth of the gold rush. To help meet the demand, cattle numbers were increased from about 250,000 in 1850 to about 1 million in 1860. A shift in cattle population centers also occurred and about 40 percent were located in the Sacramento and San Joaquin valleys. By 1862, the number of cattle in the State was about 3 million head.

The first flock of sheep was brought to San Diego in 1770. In 1783, the sheep numbered 188 head. By 1831, however, sheep at the missions numbered 153,455 head. Although sheep were important to the Spanish and Mexican economies of California, intensive labor requirements and predation problems tended to keep the flocks at minimum levels.

With the gold rush, local sheep were used to feed the miners, and large numbers of sheep were imported. Some 551,000 head, for example, were brought in from New Mexico from 1852 to 1860. As the mining boom declined, attention was turned from mutton production to wool production. Sheep numbers peaked in the 1870's with about 6.4 million head and, for a time, California led the Nation in sheep and wool production.

Damage to Resources

Forage needs precipitated the period of overstocking and overgrazing mountain meadows. High precipitation and floods in 1861-62 were followed by drought in 1862-63 and 1863-64. It was during these and subsequent drought periods, such as 1876-77, that the practice of summer grazing in the mountains began.

Cattle may have grazed the mountain meadows as early as 1861. In 1864, some 4,000 head were reported on the Big Meadow Plateau (King 1902), which is part of the Sequoia National Forest.

Sheep grazing rapidly became the dominant use of the meadows. Tales of damage to resources were to become legendary. Evidently, few, if any, passes between meadows in the Sierra Nevada were not crossed by flocks in search of greener pasture. Sheep grazing was blamed for reduced wildlife populations. Not only did sheep consume forage, which would have been available for wildlife, but diseases carried by sheep were transmitted to wildlife, particularly bighorn sheep *(Ovis canadensis)*. Also, herders took a toll of native predators in defense of their flocks.

Early efforts by the National Parks and Forest Reserves to regulate grazing were mostly fruitless. Within the National Parks, control was gradually gained, and the period of heavy sheep use was over by 1900. In the Forest Reserves, however, effective control did not begin until after the National Forest System in the Department of Agriculture was established in 1905-06 and grazing permits were required.

After the peak period, sheep grazing declined in significance. And today, cattle have all but replaced sheep in the National Forests. Except for a few situations associated with private holdings, grazing for livestock production is not allowed in the National Parks. From the 1920's into the 1970's, grazing for cattle production generally declined on the National Forests. That trend has now reversed in response to need and the mandate for public land managers to increase red meat production. Nevertheless, pressure from special interest groups to reduce or to eliminate livestock grazing as a forest use continues.

Although grazing of meadows for livestock production has declined, other uses have increased. The large pack trips of the early 1900's, with as many as 100 head of stock, are gone. But considerable horse and mule grazing still occurs, and in some areas is considered a problem. Use of meadows and meadow edges by backpackers has dramatically increased. Evidence of people damage to meadows and surrounding vegetation is increasing. Such damage is observed on bands of meadow around high-elevation lakes where trampling by fishermen has made trails.

Abuse has so damaged the meadow resource that general, widespread deterioration of mountain meadows in California has been indicated (Bailey and Connaughton 1936). For Sequoia and Kings Canyon National Parks, meadow deterioration has been well documented (Bennett 1965, Sharsmith 1959, Sumner 1941, Vankat 1970). Nevertheless, little effort was expended to study and understand meadows of the Sierra Nevada until the 1960's.

In June 1965, the Pacific Southwest Forest and Range Experiment Station and Sequoia and Kings Canyon National Parks started a joint meadow research program with a study of Funston Meadow on the Kern River (Hubbard and others 1965). After completing that study, they turned their attention to meadows of the Rock Creek drainage southwest of Mount Whitney in Sequoia National Park. Work was done by the University of California under a cooperative agreement with the Station, and was funded by the National Park Service. That research continued until 1972.

Since then, the Station has continued to study mountain meadows through its Range Research Work Unit, headquartered in Fresno. The National Park Service has continued to support meadow research by universities and individuals. The Sierra Club sponsored a wilderness impact study (Stanley and others 1979).

CLASSIFICATION OF MEADOWS

Meadows in the Sierra Nevada of California are wetlands or semiwetlands supporting a cover of emergent hydrophytes and mesophytes and dry herbland of the subalpine and alpine zones (Ratliff 1982). They may be classified in a number of ways, including by wetness, range type, altitude, physiography, vegetation, and sites. A single classification is usually applied to an entire meadow, with little if any recognition given to different sites. Wet, moist, or dry meadows can be further classified, however, by range type, which indicates the vegetation type and dominant species (U.S. Dep. Agric., Forest Serv. 1969).

Meadows of Gaylor Lake Basin in Yosemite National Park were classed as wet, moist, and dry (Klikoff 1965). The wet type is the shorthair *(Calamagrostis breweri)* type of Sumner (1941), and the dry type corresponds to the short-hair sedge *(Carex exserta)* type described by Bennett (1965). Similarly, in Yosemite Valley, meadow sites were classified in relation to a drainage gradient (Heady and Zinke 1978). Sedges (*Carex* species) decreased in percentage of species composition from depression bottoms to sites excessively drained. Grasses generally increased in the composition, becoming more prevalent than sedges on the better drained sites.

Kings Canyon National Park meadows have been classified into wet, woodland, and shorthair types (Sumner 1941). The wet meadow type consists of sphagnum, coarse-leaved sedge, fine-leaved sedge, and grass subtypes, and division of the woodland meadows into broad-leaved and coniferous subtypes was suggested (Bennett 1965).

Meadows have been classed as montane (midaltitudinal) and subalpine and alpine (high altitudinal) (Sharsmith

1959). Three classes—level meadows, hanging meadows, and stringer meadows—have been used along Rock Creek in Sequoia National Park (Harkin and Schultz 1967).

A physiographic classification with two classes in the top category was proposed (Benedict 1981). The classes were meadows with vegetated margins (type 1) and meadows with sandy margins (type 2). A lower level category contained topographic classes (basin, slope, and stream), and within these were classes of a geological category. The classification, like most others, dealt with meadows as the individual units to be classified.

At the Blodgett Forest Research Station in California, rush *(Juncus)*, forb, and dry grass meadow vegetation classes were identified (Kosco 1980). A fourth class contained rushes, sedges, grasses, and forbs in about equal proportions.

Sites within meadows are considered individual units (Ratliff 1979, 1982). Fourteen meadow site classes are described on the basis of current vegetation. The classes are viewed as approximations to series within meadow subformations (Hall 1979).

In developing the classification scheme provided in this guide, I have drawn from the authors cited, classifications of meadows from other areas, and my experience. The scheme follows the basic concepts of Hall (1979). The basic units of classification are meadow sites. I have defined a meadow site as an area of meadow with a homogeneous species composition and a general species composition different from that of adjacent areas (Ratliff 1979). Major categories of the system are subformations, series, and associations.

Although the classification is still largely heuristic, I believe it will serve as a vehicle for communication between land managers and between managers and researchers.

Subformations

A subformation is an aggregation of series with a given physiognomic character. Meadow sites are classified by subformations which, in turn, are divided into series and associations. The physiography of a meadow defines its subformation. Categories used to describe the physiography are margin type and topographic position (Benedict 1981) and plant belt (Sharsmith 1959). Codes for the classes here and in following sections are provided for use on maps, data records, or where more detailed descriptions are not required:

Category:	Class	Code
Margin type	Vegetated	A
	Sandy	B
Plant belt	Subalpine	A
	Montane	B
Topography	Basin	A
	Slope	B
	Stream	C

With this system, 12 broad meadow subformations are possible. If we use the letter codes above, the six subformations with vegetated margins are: A-A-A (vegetated margin-subalpine-basin), A-A-B (vegetated margin-subalpine-slope), A-A-C (vegetated margin-subalpine-stream), A-B-A (vegetated margin-montane-basin), A-B-B (vegetated margin-montane-slope), and A-B-C (vegetated margin-montane-stream). The six subformations with sandy margins are the same as above, except that the first code will be a B, meaning sandy margin.

Vegetated margin meadows have a herbaceous cover that extends unbroken, except by rocks or trees to or into the timbered slopes. Sandy margin meadows have a zone of sand and gravel that separates areas of meadow from each other or from the timbered slopes. The montane and subalpine plant belts in the Sierra Nevada extend from lower elevations of 1,968-4,921 ft (600-1,500 m) to upper elevations of 9,842-11,483 ft (3,000-3,500 m) (Rundel and others 1977). They include lodgepole pine forest as part of the "upper montane" zone. I include the upper reaches of lodgepole pine forest in the subalpine, however, and so I more closely follow Storer and Usinger (1963). True alpine meadows have not been included in my studies and, therefore, the alpine belt is not included at this time. Basin meadows are formed in old lakes or behind terminal moraines. Streams passing through them will usually have distinct meanders. Slope meadows are formed below seeps or springs, and they may or may not be strongly sloping. Stream meadows, formed along either permanent or intermittent streams, are commonly called stringer meadows.

Series

A series as used here is a group or cluster of meadow sites that have the same kind of margins, occur in the same plant belt, occupy equivalent topographic positions, and possess similar vegetative and hydrologic properties. Within subformations, series can be defined according to hydrological properties and vegetation. I view the hydrologic and vegetative categories as being on the same level of abstraction. Either alone or combined they may be considered to define meadow series.

Hydrology

There are 72 theoretical hydrologic series (2 by 2 by 3 by 6).

The six hydrologic classes are:

- (A) raised-convex—a site (with an enclosed open water surface) occuring as a mound above the surrounding meadow.
- (B) hanging—a site that occurs on a slope and is constantly watered by flows from springs and seeps.
- (C) normal—a site that obtains water from the water table, is recharged by precipitation, and may dry in the surface during summer.

• (D) lotic—a site that is characterized by moving water and constantly watered by flows from upstream.

• (E) xeric—a site that occurs on a slope or bench, is seasonally recharged by precipitation, and becomes quite dry during summer.

• (F) sunken-concave—a site that is characterized by ponded water and is seasonally recharged by flows from upstream.

The hydrologic category has its basis in an intuitive consideration of meadow hydrology and an adaptation of the classification described by Gosselink and Turner (1978).

Hydrology of a meadow or meadow site is the variable controlling potential vegetation. Chemistry of the incoming water affects the nutrient availability and cycling. Velocity of the flows affects the particle size of materials transported to and from a meadow. Seasonality and reliability of the flows largely determine the vegetational stability. Alteration of a meadow's hydrology will change its species composition. Only by maintaining or reestablishing the natural hydrology of a meadow or meadow site is it possible to maintain or recover its potential or stable-state vegetation.

The concept of a hydrologic classification of meadows or meadow sites is certainly not new (Hall 1979; Harkin and Schultz 1967; Heady and Zinke 1978; Hormay 1943; Klikoff 1965; Sumner 1941; U.S. Dep. Agric., Forest Serv. 1969). The classification of Gosselink and Turner (1978), however, is probably the first to use the dynamics of meadow hydrology for classification. They used four groups of characteristics to describe the classes: water inputs, type of flow, water outputs, and hydro-pulse. I have added xero-pulse to their list, added or removed some individual characteristics, and revised the classes to fit conditions of the Sierra Nevada. Also, I have attempted to judge the significance of each characteristic to each hydrologic class.

Input Flows—The sources of water for meadow sites are input flows *(table 1)*. Capillarity is considered to be a major source for the raised-convex (bog) class (Gosselink and Turner 1978). Although capillary water is needed for sites of that class in the Sierra Nevada, hydrostatic water is of equal or greater need. Development of subsurface aquifers within a meadow system has been discussed (Leonard and others 1969). Water carried in a layer of coarse sediment between layers of peat may surface downslope. At that point, peat deposits may build up, restricting the flow of water. A raised mound with an open water surface above the water table of the surrounding meadow develops to give a site of the raised-convex class. Such aquifers also supply water to other parts of a meadow.

Hydrostatic flows from springs and seeps are the main water sources for hanging meadow sites. The water is either forced to surface by bedrock or emerges from the base of lateral moraines (Benedict 1981). On peat, much of the water movement is at or near the surface (Heinselman 1970). Down slope from the water source, therefore, the older more compressed peat may act to prevent downward flow of the water. The peat layers of aquifers are less porous than the coarse material (Leonard and others 1969). And, in my studies of the Sierra Nevadan meadows, relatively dry soil frequently has been found below a water saturated surface, especially below a seep or spring.

Precipitation is necessary for all meadow sites, but more directly so to some than to others. Normal meadow sites obtain water mainly by capillarity from the water table. They depend, however, on precipitation, upslope flows, or both, to recharge the upper soil layers with water. Xeric

Table 1—*Hydrodynamic classification of meadow sites in the Sierra Nevada, California*[1]

Classification variables	Hydrologic class: Raised convex	Hanging	Normal	Lotic	Xeric	Sunken concave
Input flows						
Capillary	*		*			
Hydrostatic	*	*		o		
Precipitation	o	+	+	+	*	+
Upstream or upslope			+	*	+	*
Internal flows						
Capillary	*	+	+			
Subsurface	*	*	*	*	*	*
Surface sheet		+	o	*	*	+
Surface rill		o	+		+	
Output flows						
Evapotranspiration	*	*	*	*	*	*
Percolation	o		*		+	o
Downstream runoff		*	o	*	*	
Hydro-pulse						
Seasonal			*	*	*	*
Seasonal/constant	*	*		+		
Xero-pulse						
Seasonal			*		*	*
None	*	*		*		

[1]Key: o = minor importance, + = moderate importance, and * = major importance.

sites are recharged mainly by precipitation, but upslope flows may influence them. Xeric sites occur on slopes and benches and may be subjected to inundation during snowmelt from barren, rocky areas above.

Lotic and sunken-concave meadow sites depend more upon upstream water than on-site precipitation. In the Sierra Nevada, such sites are usually found in topographic basins or flooded areas along streams. Lotic sites are characterized by moving water. Velocity and depth of water during spring runoff affect their species compositions. Sunken-concave sites are characterized by ponded water. Their species compositions are influenced by the depth of ponding after spring runoff.

Internal Flows—Those flows, occurring once water reaches a site, are internal flows. Subsurface flows occur on all sites. Maintaining the water table by subsurface flows is of major importance to all meadow sites. Capillary rise of water is important to both the raised-convex and the hanging meadow classes. In an invasion of "bog forest" by sphagnum moss, the sphagnum advanced uphill into climax forest to a height of 15 ft (4.5 m) above the bog (Cooper 1912). Water held in the capillary pores of sphagnum peat is necessary for such a phenomenon to occur. Mosses of some kind are usually found in hanging and raised-convex meadow sites.

The surface sheet type of internal flow is represented by surface water of variable depth covering the entire site. The water is usually moving. This movement is especially apparent in normal, lotic, and xeric meadow sites. Except for overflow, water is stagnant on sunken-concave sites. Water depth at flowering time is a key variable controlling species composition (Hormay 1943). Depth of the surface sheet flows, therefore, primarily determines the species of lotic sites, as does depth of standing water on sunken-concave sites.

Surface rill flows are akin to rill washing (Gustafson 1937). After a rainstorm, rill flow is evidenced as shallow puddles in small depressions or slight movements of litter. Rill flows can be valuable to hanging meadow sites, especially when surface water is gone. Except for the usually dry summers in the Sierra Nevada, rill flows would be of major significance on normal and xeric meadow sites.

Output Flows—Water lost or removed from a meadow site constitutes an output flow. Although evapotranspiration occurs regardless of the kind of meadow site, as an output flow it is of greatest significance on raised-convex and sunken-concave sites. In both, losses to percolation are relatively minor because of the presence of head pressure on raised-convex sites and an impermeable or slowly permeable layer on sunken-concave sites. Because both are ponded, downstream runoff does not occur.

Percolation flow is important on normal and xeric sites. On normal meadow sites, water may percolate to the water table and eventually emerge downstream. A generally shallow soil means that deep percolation will not occur on xeric sites, except when water enters fissures in the underlying rock. Thus, although percolation occurs, the soil is quickly saturated and most output becomes downstream runoff.

Downstream runoff is a major output flow of hanging and lotic meadow sites. Owing to the saturated or slowly permeable nature of the underlying materials, percolation is slow.

Hydro-pulse—The regularity of additions of water to the meadow site system is reflected by its hydro-pulse. Conditions of fairly constant moisture regimes from year to year have been considered the key to developing and maintaining meadows in northeastern California (Hormay 1943). The same may be assumed for meadows of the Sierra Nevada.

Although subsurface flows may continue throughout summer in the Sierra Nevada, water input is mainly snowmelt and spring and early summer rains. Normal, lotic, xeric, and sunken-concave meadow sites largely depend upon those inputs. Raised-convex and hanging sites are also affected by seasonal regularity of snow and rain. After the flush of water early in the season, a fairly constant input of water is maintained. Depending upon its location in the watershed, size of the watershed, and mount of precipitation, lotic sites may also receive rather constant additions of water through the summer.

Xero-pulse—The regularity with which a meadow site dries is reflected in its xero-pulse. Raised-convex, hanging, and lotic sites have high soil water-content, except in the situation of extreme prolonged drought over a number of years. Soils of xeric sites are usually quite dry by about the first of August. Having deeper soils and possible access to the water table, normal meadow sites usually have some moisture at depth. They may, however, be fairly dry in the surface layer. Surface water is usually gone from sunken-concave sites by mid-August, and the soil may dry to considerable depth before fall.

Vegetation

The 14 vegetation classes (Ratliff 1979-1982) and the 19 associations (Benedict 1981) have been combined to give 21 vegetative series *(table 2)*. These are general classes intended to reflect species abundance in the community. They are based on current, rather than potential or climax, vegetation. The list of series may be enlarged or condensed as new information is brought into the system.

Associations

Associations are composed of sites of the same hydrologic and vegetative series and have close floristic similarity. They comprise the lowest category of classification. The most extensive and intensive work to define meadow associations in the Sierra Nevada has been that of Benedict (1981).

The same species are expected to grow together on sites having similar environments. Nonsalient differences in the environments, however, can affect the species present, their abundance, or both. As a result, differences between associations of the same series may not be immediately obvious, and intensive study is usually necessary to identify different associations. For most administrative purposes, therefore, classification of meadow sites to the association level would likely

Table 2—*Vegetative series and associations of meadow sites in the Sierra Nevada, California*

Series		Association	
Code	Name	Code	Name
A	*Agrostis* (Bentgrass)		
B	*Artemisia rothrockii* (Rothrock sagebrush)	1	*Artemisia rothrockii*
C	*Calamagrostis breweri* (Shorthair)	1	*Calamagrostis breweri—Oryzopsis kingii*
		2	*Calamagrostis breweri—Aster alpigenus*
		3	*Calamagrostis breweri—Vaccinium nivictum*
		4	*Calamagrostis breweri—Trisetum spicatum*
D	*Calamagrostis canadensis* (Bluejoint reedgrass)	1	*Calamagrostis canadensis—Dodecatheon redolens*
E	*Carex exserta* (Short-hair sedge)	1	*Carex exserta*
F	*Carex heteroneura*	1	*Carex heteroneura—Achillea lanulosa*
G	*Carex nebraskensis* (Nebraska sedge)		
H	*Carex rostrata* (Beaked sedge)	1	*Carex rostrata*
		2	*Carex rostrata—Mimulus primuloides*
I	*Deschampsia caespitosa* (Tufted hairgrass)	1	*Deschampsia caespitosa—Cardamine breweri*
		2	*Deschampsia caespitosa—Senecio scorzonella*
		3	*Deschampsia caespitosa—Senecio scorzonella—Achillea lanulosa*
J	*Eriogonum* (Buckwheat)	1	*Eriogonum—Oreonana clementis*
K	*Gentiana newberryi* (Newberry gentian)		
L	*Heleocharis acicularis* (Slender spikerush)		
M	*Heleocharis pauciflora* (Fewflowered spikerush)	1	*Heleocharis pauciflora*
		2	*Heleocharis pauciflora—Mimulus primuloides*
N	*Hypericum anagalloides* (Tinkers penny)		
O	*Juncus* (Rush)	1	*Juncus orthophyllus*
P	*Muhlenbergia filiformis* (Pullup muhly)		
Q	*Muhlenbergia richardsonis* (Mat muhly)	1	*Muhlenbergia richardsonis*
R	*Penstemon heterodoxus* (Heretic penstemon)	1	*Penstemon heterodoxus—Achillea lanulosa*
S	*Poa* (Bluegrass)		
T	*Trifolium longipes* (Longstalk Clover)		
U	*Trifolium monanthum* (Carpet Clover)		

be impractical. When known, however, the association to which a site belongs should be identified as part of its classification.

Classification to the association level may be required to assess range condition and trend in condition and for some research programs. Because the associations are based on current vegetation, identification of the association present on a site may provide a clue to its potential. A site of vegetative series C (shorthair) with association 2 *(Calamagrostis breweri—Aster alpigenus)* may represent a degraded stage from association 1 *(Calamagrostis breweri—Oryzopsis kingii)* of that series, for example. And, perhaps, a change in the association of a site could even be taken as an indication of the presence of trend in condition.

Meadow Soils

Development

Soil moisture regime is the single most significant property that determines the existence and characteristics of a meadow (Wood 1975). The characteristics of a meadow depend upon consistency in the moisture regime from year to year (Hormay 1943). Situations favorable to accumulating fine-textured materials and establishing shallow water tables are prerequisite to meadow development. A fine-textured soil is needed to draw water to shallow rooted meadow plants by capillary rise. A shallow water table is needed so that water does not have to be raised too far. A favorable situation has (1) a relatively impervious bedrock floor; (2) an upper drainage area of sufficient size to supply seepage to maintain a shallow water table well into the growing season; (3) a gentle gradient; and (4) a favorable drainage area-to-slope relationship, perhaps the most valuable characteristic.

Drainage area above a meadow affects the volume of annual water input. Slope affects water velocity, sediment load, and deposition. Above a critical slope threshold, an alluvial valley floor is usually unstable (Schumm 1977). As valley floor slope increases so does valley instability. Meadows with drainage areas larger than 512 acres (207.2 ha) and with slopes greater than two percent are likely to be unstable (Wood 1975). Usually, such meadows have well-defined stream channels through them. Meadows with drainage areas smaller than 512 acres or with slopes less than two percent, or both, are likely to be stable. Meadows with gentle slopes and small drainage areas do not usually have well-defined stream channels. Steeply sloped meadows with small drainage areas tend to contain straight stream channels. Gently sloped meadows with large drainage areas tend to contain sinuous stream channels. Large drainage areas with steep stream gradients do not contain meadows.

Peatification and Marshification—Some meadows of the Sierra Nevada have developed through the typical successional pattern from glacial or montane lakes. Many Sierra

Nevadan meadows, however, were created abrubtly because of changes in the moisture regime (hydrology) brought about by major changes in climate (Wood 1975). Others have developed by aggrading of surfaces and lifting of the water table with the land.

Bodies of water transform into peat in the process of being filled with peaty materials due to the growth of vegetation in them. The materials are deposited as strata, with the type of material deposited related to water depth and the organisms present. This process is called peatification (Vilenskii 1957). The five steps in the process are:

1. Deposition of lake marl—lake silt rich in lime;
2. Deposition of sapropelite—sedimentary peat consisting mainly of minute animal and plant remains;
3. Deposition of remains of rooted shore vegetation—the type of material deposited largely depends upon the depth of water;
4. Gradual reduction in depth and surface area, with zones of shore vegetation moving farther out;
5. Finally, the complete filling of the pond or lake with plants and peat and the development of a marsh.

The peatification process may follow different routes. Under favorable conditions vegetation may grow out over the surface of the water forming a raft upon which other plants grow. Materials dropping from the bottom of the raft settle to the bottom. At the same time, the raft becomes thicker. Peat materials, therefore, build up from both the top and the bottom. This is probably the manner in which sites of the raised-convex hydrologic class are gradually filled.

Another process through which peat accumulates is marshification, or the swamping of dry lands, which occurs at some northern and moderate latitudes (Vilenskii 1957). This process is driven by a rise in the water table because of decreased evapotranspiration. A weed stage generally follows, as after the harvest of timber, and lasts for one year or so. The weed stage is succeeded by a meadow stage composed largely of rhizomatous grasses. After about two years in the meadow stage, sphagnum moss begins to invade the meadows. Mosses other than sphagnum may first become established in the grass meadow providing conditions more suitable for sphagnum. Alternately, reedgrasses (*Calamagrostis* spp.) and sedges become established and these are followed directly by sphagnum.

Meadows have sometimes developed after logging in the Sierra Nevada. Conditions favorable for marshification occur where surface runoff is poor, where the parent material is from rock types with mineral compositions favorable to clay formation, where subsurface drainage is poor, and where fine-textured material that effectively prevents or slows the outflow of seepage water is deposited.

Poor subsurface drainage may result from a compact, impermeable illuvial B horizon or by bedrock. But an irreversible loss of moisture from peat (Pons 1960, Schelling 1960) during periods of drought or because of draining may produce a slowly permeable layer that acts to keep incoming water at or near the surface. These two conditions combine to keep water at or near the surface. On sites of the hanging hydrologic class, drought and drainage frequently produce the anomaly of a relatively dry soil beneath a wet meadow surface. These same two conditions may also permit marshification in uncut forest—especially near existing marshes. The invasion of the "bog forest" by sphagnum moss has been described (Cooper 1912). The sphagnum advanced uphill into climax forest to a height of 15 ft (4.5 m) above the original level of the marsh. The process of marshification appears to be active around some seeps and springs in the Sierra Nevada, but the particular mosses involved may not be sphagnum.

Stratigraphy and Chronology—Four main depositional units are recognized in montane meadows of the Sierra Nevada (Wood 1975). A unit of coarse pre-Holocene alluvium several feet thick rests upon the bedrock. The next unit is a paleosol (old soil), which shows profile development and in some meadows a gleyed (intensely reduced, with ferrous iron and neutral gray colors) condition. This buried soil unit extends into the alluvium and in places to the bedrock. It was developed between 8,705 and 10,185 years B.P. (before present). The third unit is composed of stratified sand deposits, dated from 8,700 to 1,200 B.P. and as much as 20 ft (6.1 m) thick. It represents a period of forest development with well-drained soils. Evidence of profile development was not found, implying geologic instability. The fourth depositional unit is 2 to 12 ft (0.6 to 3.7 m) thick and is made up of interstratified layers of sorted grus, peat, and organically rich sandy loams. This unit represents development of open, wet meadows and has been deposited since 2,500 years B.P. Repetition of sand-sod-peat layers has effectively prevented soil profile development. Continued geological instability in the drainage area of a meadow is indicated. Present sand and gravel depositions on meadows, therefore, may be continuations of a natural process.

I have frequently observed the fourth depositional unit described by Wood (1975). One profile, an extreme situation perhaps, had 10 separate well-delimited layers in the first foot (30 cm) of soil. But not all montane meadows of the Sierra Nevada have the four soil unit sequence. Some meadows have well-developed soils and have evidently been stable for up to 10,000 years. One study concluded that "meadow ecosystems are as stable as the surrounding vegetation" (Benedict 1981, p. 80). What occurs on the drainage area above it, therefore, greatly affects what occurs on a meadow.

Many subalpine meadows of the Sierra Nevada are not infilled glacial lakes, and differ somewhat in their stratigraphy from montane meadows (Wood 1975). Glacial till, fluvial deposits of gravel and sand, and a topsoil comprise the basic stratigraphic sequence. Accumulation of topsoil at Tuolumne Meadow, Yosemite National Park, has occurred during the last 2,300 years. On glacial till hummocks, the beginnings of genetic soil horizon development occurs and A, B, and C horizons can be recognized. Topsoils of low areas range to 3 ft (0.9 m) thick. In some, thin sandy lenses indicate minor, but continued, deposition of materials from the drainage area.

Bands of meadow around high-elevation lakes, as hypothesized, developed through growth of moss and Sierra bilberry *(Vaccinium nivictum)*, alpine laurel *(Kalmia polifolia*

microphylla), or both, over and around boulders at waters edge (Ratliff 1973). The shrubs serve to anchor the moss. In turn, the moss acts to raise the water table and support extension of shrub branches between boulders. This shrub-moss base (with trapped dust and sand) constitutes the parent material of the soil. Eventually, other plants become established and a more typical A horizon is produced.

One, and often two, layers of a white pumiceous volcanic ash (tephra) were found in several meadow soil profiles examined by Wood (1975). Source of the tephra was found to be the Mono-Inyo craters. The more recent layer was named "tephra 1" and the older layer "tephra 2." Times of deposition, determined by radiocarbon techniques, were 720 years B.P. for tephra 1 and 1,200 years B.P. for tephra 2 (Wood 1975).

Presence of these tephra layers and their ages are significant. They serve as check points for estimating the deposition rates of materials above them. About 10 inches (25 cm) of meadow soil have been deposited above tephra 1, and about 22 inches (56 cm) of meadow soil have been deposited above tephra 2 (Wood 1975, fig. 3-29). Those amounts of soil translate into 1.4 inches (3.4 cm) per century for tephra 1 and 1.8 inches (4.7 cm) per century for tephra 2. For the 480 years between 1,200 and 720 B.P., the rate of accumulation was about 2.5 inches (6.5 cm) per century. It appears that the rate of accumulation has slowed since tephra 1 was deposited. Erosion or decomposition rates, or both, however, may have accelerated.

Classification

Little real effort has been expended toward classification of meadow soils in the Sierra Nevada. Some soils appear to be of organic origin, others are clearly of mineral origin. The earliest classification effort of which I am aware was that of the U.S. Department of Agriculture, Soil Conservation Service (1962). Four kinds of soils from 12 meadows on the Sierra National Forest were described: normal meadow, drained meadow, alluvial timber, and peat meadow. All were said to have effective depths of 3 to 5 ft (0.9 to 1.5 m). The first three overlay weathered granitic material, the peat meadow overlays decayed organic material. Surfaces of normal meadow soils vary from peat-like to silt loam in texture, have gravel to silt loam subsoil, and are slightly alkaline to moderately acid. The water-holding capacity of these soils is high, with moderately slow to very slow runoff. They are poorly to imperfectly drained with very slow to moderate permeability, and have a seasonally high water table.

Drained meadow soils have loamy, coarse sand to loam surfaces and coarse sand to silt loam subsoils. They are slightly to moderately acid and have low to moderate water-holding capacities with slow to moderate runoff. They are slowly to moderately permeable and are imperfectly to well drained.

Alluvial timber soils have slow surface runoff, moderate to rapid permeability, and are imperfectly to well drained. Their surfaces vary from coarse sandy loam to loam over coarse sandy loam to sandy loam subsoils. Surface runoff is slow, with water-holding capacity ranging from low to high. Reaction is slightly to moderately acid.

Peat meadow soils have peat surfaces and subsoils and high water-holding capacities. They generally have water within 1 ft (30 cm) of the surface and are slowly permeable and slowly drained but with slow runoff. Their reaction is slightly acid.

Recently, the soil taxonomy system (Soil Survey Staff 1975) was applied to meadows of the Sierra Nevada (Wood 1975). Meadow soils, not classed as Histosols, are considered either udic or perudic cryofluvents. *Udic* and *perudic* refer to the soil moisture regime, *cryo* refers to the soil temperature regime, *fluv* refers to water-deposited sediments, and *ent* refers to Entisol. Entisols are dominated by mineral soil materials; they do not have distinct pedogenic (related) soil horizons.

In general, a soil is considered a Histosol when more than one-half of the upper 80 cm of the soil is organic, or if organic material of any thickness rests on rock or fragmented material, the interstices of which are filled with organic materials (Soil Survey Staff 1975). The first situation is likely in infilled lakes. The "typical peaty" meadow soil (Aliev 1964) appears to generally express such conditions. The second situation is found at seeps, springs, and along some lake shores. "Dern-peaty" and "peaty-dern" soils have been described (Aliev 1964). These have thin surface layers high in organic matter.

Histosols saturated for long periods have a minimum of 12 to 18 percent organic carbon, depending upon the amount of clay in the mineral fraction (Soil Survey Staff 1975). The more clay (up to 60 percent), the more organic carbon is required for the soil to be a Histosol. Histosols not saturated for long periods have at least 20 percent organic carbon. The presence of large amounts of organic matter therefore distinguish the Histosols from the orders of mineral soils. Histosols have "histic epipedons" (a kind of diagnostic surface horizon). Certain Entisols and Inceptisols, however, also may have histic epipedons. Presence of a histic epipedon, therefore, does not automatically signify a Histosol.

Meadow soils of the Histosol order are fluvaquentic borofibrists or borohemists (Wood 1975). *Aqu* of fluvaquentic refers to water; therefore, such soils are like water-deposited wet Entisols. *Boro* soils have a frigid temperature regime. The *fibrists* are Histosols that consist largely of recognizable plant remains, which are not easily destroyed by rubbing between the fingers. The *hemists* are Histosols that are far more decomposed; most of the material is easily destroyed by rubbing. Here the water table fluctuates appreciably permitting more rapid decomposition than with fibrists.

Characteristics—Meadow soils in the Sierra Nevada cannot be characterized by soil horizon. Many of the layers in organic soils do not meet the definition of a soil horizon, and it is not always possible to distinguish true soil horizons when they are present (Soil Survey Staff 1975). It is frequently difficult to determine the boundaries between fibric, hemic, and sapric soil materials. Variation in textures frequently represent different depositional events (Hubbard and others 1966). One study found that meadow soils in Yosemite Valley did not approach maturity (Heady and Zinke 1978). And

Table 3—*Average values of some soil characteristics on 82 meadow sites of the Sierra Nevada, California by depth segment*

Soil characteristic	Depth segment		
	0 to 10 cm	10 to 20 cm	20 to 30 cm
pH	5.0	5.1	5.1
Composition (pct[1])			
Sand	54	56	58
Silt	34	31	30
Clay	11	12	13
Organic matter	21	13	10
Gravel	6	6	7

Source: Ratliff (1979)

[1]Values are for the 2-mm soil fraction, except for percent for gravels, which is calculated on the basis of bulk sample weight.

whether the soil is of organic or of mineral origin, it has been shown that true pedogenic soil horizons are rare in modern meadow soils (Wood 1975).

Sampling and describing meadow soils by depth segment seems a viable alternative. From the surface, the first 10-cm soil segment is the most heavily affected by grazing and other activities. Frequently, it is the zone of maximum root concentration and is highly organic. From 10 to 20 cm, organic matter is usually moderately reduced, but plant roots are usually abundant. Sufficient mineral matter is present for particle size determination, and the segment is affected by surface disturbances. The 20- to 30-cm segment frequently is well below the zone of maximum root concentration, is usually relatively low in organic matter, and is below most surface activity influences. Deeper segments, although significant for understanding meadow genesis, are little influenced by current management activities.

I have used and suggest use of the 10- to 20-cm depth segment as a standard for comparing and characterizing meadow sites. For study of meadow response to management activities, the top segment should also be used. The lower segments should be used when fuller understanding of meadow development is required.

Detailed and specific information on soil characteristics of meadow sites and site classes are available (Benedict 1981; Bennett 1965; Heady and Zinke 1978; Hubbard and others 1966; Ratliff 1979, 1982).

Soils of Sierra Nevada meadows generally tend to be strongly acid to very strongly acid with a pH of about 5.0 *(table 3).* A low pH of 3.4 and a high pH of 8.0 have been reported. Textures of meadow soils range from coarse sands to an occasional clay. Most soil textures are loamy sands, sandy loams, and loams. The average texture by depth segment to 30 cm is sandy loam.

Organic matter content of meadow soils (estimated by gravimetry and ignition) has ranged between 4 and 90 percent. The organic content usually decreases with depth. The average organic matter content in the 10- to 20-cm segment *(table 3)* was 38 percent less than at 0 to 10 cm. Between the middle and lower segments the rate of decrease is much less. The depositional sequence may, however, result in organic matter at depth being more than in the surface segment. In one soil, for example, organic matter contents were 25, 16, and 31 percent in the respective depth segments from the surface. The effect appears related to more sand and less silt in the 10- to 20-cm segment than in the other two.

Sequences of deposition and erosion determined where and how much gravel was in each soil, but it appears that gravels were deposited in about equal amounts in the three depth segments *(table 3).* Because the soil samples were extracted with a ¾-inch-(1.9-cm) diameter sampling tube, only pebbles of finely gravelly size (Soil Survey Staff 1975) or slightly larger were included in my samples. Nevertheless, gravelly meadow soils are frequently observed. Eleven of the 82 soils contained 15 percent or more gravels in one or more depth segments, and meadow soils with cobblestone size fragments have been reported.

Series Descriptions

A series, as used here, is a group or cluster of meadow sites that have the same kind of margins, are found in the same plant belt, occupy equivalent topographic positions, are influenced in the same way hydrologically, and have similar vegetations.

The classification presented in this paper provides for 1,512 ($2 \times 2 \times 3 \times 6 \times 21$) theoretical meadow series. Of course, not all are biologically possible. Among the 126 theoretical hydrologic-vegetative combinations, 32 have been identified in the field *(table 4).* Few, if any of these, are found under all combinations of margin type, plant belt, and topographic position. Even so, a large number of possible series are obvious.

Only in a few situations have enough sites been studied to adequately define the series. Therefore, rather than attempt to provide modal class concepts, my aim is to enable the manager to use the classification scheme. For that purpose, each series discussed is represented by a single site (one that I personally have studied). The site in each situation reflects as nearly as possible my concept of sites that belong to the series.

The code for a meadow series includes five letters representing, in order, the nature of the margin, the plant belt, the topographic location, the hydrologic series, and the vegetative series. The association to which a site belongs may be identified in the classification code by adding the association number to the vegetative series.

Series A-B-C-B-G (fig. 1) is represented in John Brown Meadow, Minarets Ranger District, Sierra National Forest. The meadow has vegetated margins, is in the montane belt at 6,791 ft (2,070 m), and is a stringer along an intermittent stream. Seeps in the upper reaches of the meadow feed the stream, but it was dry some distance below the site in mid-August 1979.

The site has a slope of 4 percent. Above the site, water is well distributed over the meadow so that there is no distinct channel. As a result, the site receives some flow from upstream. Its main input flow is from seepage at the upper side. Water flows slowly across the site to join the stream.

Table 4—*Hydrologic-vegetative series relationships common in the Sierra Nevada, California*

Vegetative series		A Raised convex	B Hanging	C Normal	D Lotic	E Xeric	F Sunken concave
A	*Agrostis* (bentgrass)		X	X			
B	*Artemisia rothrockii* (Rothrock sagebrush)					X	
C	*Calamagrostis breweri* (Shorthair)			X			
D	*Calamagrostis canadensis* (Bluejoint reedgrass)		X				
E	*Carex exserta* (Short-Hair sedge)					X	
F	*Carex heteroneura*			X			
G	*Carex nebraskensis* (Nebraska sedge)		X	X	X		
H	*Carex rostrata* (Beaked sedge)				X		
I	*Deschampsia caespitosa* (Tufted hairgrass)		X	X			
K	*Gentiana newberryi* (Newberry gentian)			X			
L	*Heleocharis acicularis* (Slender spikerush)				X		X
M	*Heleocharis pauciflora* (Fewflowered spikerush)	X	X	X			
N	*Hypericum anagalloides* (Tinkers penny)	X	X				
O	*Juncus* (Rush)			X			
P	*Muhlenbergia filiformis* (Pullup muhly)		X	X			
Q	*Muhlenbergia richardsonis* (Mat muhly)					X	
R	*Penstemon heterodoxis* (Heretic penstemon)			X			
S	*Poa* (Bluegrass)			X		X	
T	*Trifolium longipes* (Longstalk clover)		X	X			
U	*Trifolium monanthum* (Carpet clover)		X	X			

Figure 1—Meadow series vegetated margin (A), montane (B), stream (C), hanging (B), and Nebraska sedge (G) in John Brown Meadow, Minarets Ranger District, Sierra National Forest.

Table 5—*Species composition for nine meadow sites of the Sierra Nevada, California*

Classification variable	Site classification								
Margin type	A	A	A	A	A	A	A	A	A
Plant Belt	B	B	B	A	B	A	B	B	A
Topographic	C	A	A	A	B	A	B	A	B
Hydrologic series	B	C	D	D	E	C	B	F	E
Vegetative series	G	G	G	H	S	C	N	L	E
Species	Percent composition[1]								
Agrostis lepida								5.7	
Aster alpigenus			0.3			12.0			
Aster occidentalis		15.3							
Calamagrostis breweri						36.3			1.0
Carex athrostachya		6.7						0.7	
Carex exserta									80.3
Carex nebraskensis	81.7	54.3	55.3						
Carex rostrata		0.3		57.3				T	
Carex simulata	6.3	0.7	36.3			T			
Dodecatheon jeffreyi			2.3				8.3		
Gentiana newberryi			0.3			5.7			
Heleocharis acicularis	2.3	1.3	16.0					69.7	
Heleocharis palustris							4.3	12.7	
Heleocharis pauciflora	2.7	3.3	7.0				0.3		
Horkelia fusca capitata					14.7				
Hypericum anagalloides			0.7			T	19.0		
Juncus nevadensis				6.0					
Lupinus breweri bryoides									5.3
Marsilea vestita								10.0	
Muhlenbergia filiformis			13.0			1.3	11.7		
Oxypolis occidentalis							19.3		
Perideridia species		6.0					T		
Phalacroseris bolanderi							8.0		
Poa pratensis					60.7				
Senecio species						7.3			
Solidago canadensis					6.7				
Solidago multiradiata						9.0			1.7
Stipa occidentalis					9.0				
Trifolium longipes		9.7							
Vaccinium nivictum						10.7			

[1]Percent composition is calculated on the basis of nearest shoot-to-point frequency. T = Trace.

Here, water depth is not enough to permit a lotic classification. And although the slope is considerably less than usual for hillside bogs, the site must be assigned to the hanging hydrologic series.

Nebraska sedge *(Carex nebraskensis)* is the vegetative series, and the species makes up 82 percent of the composition here *(table 5)*. Analogne sedge *(Carex simulata)* is the next most frequent species on the site.

Organic matter content of the soil (26 percent) is high *(table 6)*. The soil texture is a sandy loam with pH 5.6. A recent overwash of sand on the site is evidenced by a high (95 percent) basal frequency of bare soil.

Some similar sites may have much less Nebraska sedge. The amounts of Nebraska sedge may not be enough to keep such sites in that vegetative series. It will likely be replaced by the fewflowered spikerush *(Heleocharis pauciflora)* vegetative series. Such a result may be a response to overuse, and such sites may have a Nebraska sedge potential.

Series A-B-A-C-G is represented by a site in Cannell Meadow, Cannell Meadow Ranger District, Sequoia National Forest *(fig. 2)*. Cannell Meadow is a large montane, basin meadow at 7,600 ft (2,316 m). A few bare spots occur where the margins are not vegetated. These may be people-caused, however, and the meadow is classed as having vegetated margins.

The site is in the normal hydrologic series. Although it is subject to sheet flows in the spring, depth of the water does not approach that of lotic sites. The water table may be at considerable depth by mid- to late summer. Nebraska sedge is the vegetative series and on this site makes up 54 percent of the species composition. Western aster *(Aster occidentalis)* and longstalk clover *(Trifolium longipes)* are the next most frequent species.

Soil on the site is a sandy loam with pH 5.7 and 8 percent organic matter. Bare soil makes up about 6 percent of the surface, and 87 percent is covered by litter. Less than 2 percent of the surface cover is moss.

Species composition on sites of this series appears to be influenced considerably by the regime of grazing. A fence separated this site (which is generally used late in the grazing season) from a site of the same series that is grazed season long. On that site, Nebraska sedge makes up 24 percent,

Table 6—*Surface cover and soil properties on nine meadow sites of the Sierra Nevada, California*

Classification variable	Site classification								
Margin type	A	A	A	A	A	A	A	A	A
Plant belt	B	B	B	A	B	A	B	B	A
Topographic	C	A	A	A	B	A	B	A	B
Hydrologic series	B	C	D	D	E	C	B	F	E
Vegetative series	G	G	G	H	S	C	N	L	E
Surface cover	Percent cover[1]								
Soil	94.7	5.7	26.7	63.0	21.3	8.0	19.0	43.0	34.0
Litter	0.3	87.3	59.7	34.0	73.7	77.3	28.3	53.7	38.7
Gravel	1.0				0.3				13.0
Rock									1.7
Wood							0.7	1.3	
Moss	1.3	1.7	9.0	0.3		2.7	50.0		
Higher plants	2.7	5.3	4.6	2.7	4.7	12.0	2.0	2.0	12.6
Soil property[2]									
Texture (Percent)									
Sand	57.1	61.0	43.2	20.6	52.8	72.4	62.0	19.8	56.4
Silt	35.4	31.9	47.6	39.7	28.9	16.6	32.2	42.2	33.1
Clay	7.5	7.1	9.2	39.7	18.3	11.0	5.8	38.0	10.5
Organic matter (Pct)	26.5	8.0	22.2	33.3	9.5	4.7	28.9	5.8	7.4
Acidity (pH)	5.6	5.7	5.0	5.2	5.5	4.9	5.2	6.0	5.1

[1]Percent surface cover is calculated on the basis of frequency of actual basal point contacts.
[2]Soil values are for the 10- to 20-cm depth segment.

Figure 2—Meadow series vegetated margin (A), montane (B), basin (A), normal (C), and Nebraska sedge (G) in Cannell Meadow, Cannell Meadow Ranger District, Sequoia National Forest.

straightleaf rush *(Juncus orthophyllus)*, 19 percent, and fewflowered spikerush, 17 percent of the composition. Also, litter was reduced to 65 percent of the surface, and soil was increased to 9 percent and moss to 24 percent.

Series A-B-A-D-G is represented by a site in Jackass Meadow, Minarets Ranger District, Sierra National Forest *(fig. 3)*. Jackass Meadow has vegetated margins, lies at 7,000 ft (2,134 m) in the montane belt, and is a basin meadow. The site is lotic—water flows over it with some depth and velocity, and the water table is almost always at or near the surface. The vegetative series is Nebraska sedge, which makes up 55 percent of the species composition. Other species of significance on the site are slender spikerush *(Heleocharis acicularis)*, fewflowered spikerush, and pullup muhly *(Muhlenbergia filiformis)*. Some Jeffrey shooting-star *(Dodecatheon jeffreyi)* and Bolanders clover *(Trifolium bolanderi)* are present. Frequently, longstalk clover is the main clover on these sites.

The soil is a loam with 22 percent organic matter and pH 5.0. Litter covers 60 percent, soil, 27 percent, and moss, 9 percent of the surface.

Figure 3—Meadow series vegetated margin (A), montane (B), basin (A), lotic (D), and Nebraska sedge (G) in Jackass Meadow, Minarets Ranger District, Sierra National Forest.

Water depth and velocity are the main variables controlling species composition of sites in this series. A depth of surface flow in spring of 10 to 15 cm is usual. On one site of this class, flow velocity for 30 consecutive days during spring runoff averaged 0.36 ± 0.04 ft sec^{-1} (11.0 ± 1.2 cm sec^{-1}). Where water is less deep, sites of this class tend to merge with sites of the tufted hairgrass *(Deschampsia caespitosa)* series. Where water is deeper and velocities are reduced, sites of this class tend to merge with sites of the beaked sedge *(Carex rostrata)* series.

Sky Parlor Meadow *(fig. 4)* on Chagoopa Plateau, Sequoia National Park, contains a representative of *series A-A-A-D-H.* The meadow has vegetated margins, lies at 8,976 ft (2,736 m) in the subalpine belt, and is in a basin.

The hydrologic series of the site is lotic. Usually surface water is found on the site even in September, and the water table is always above or near the surface. Although I have not measured flow velocity, it should be less and water depth should be greater than on lotic Nebraska sedge sites.

The vegetative series is beaked sedge and constitutes 57 percent of the composition. Analogne sedge is a codominant, and Nevada rush *(Juncus nevadensis)* is frequently seen.

Soil texture is clay loam with 33 percent organic matter and pH 5.2. Because of fairly rapid decomposition at the site (Ratliff 1980), the amount of soil (63 percent) is almost twice that of litter (34 percent).

In the subalpine belt, beaked sedge sites tend to merge with open water as water becomes deeper and with grass dominated vegetative series as water becomes more shallow. Montane sites with the same hydrologic and vegetative series tend to merge with those of ephemeral-lakes in deeper water and with those of the Nebraska sedge series in more shallow water.

Crane Flat Meadow in Yosemite National Park contains an example of *series A-B-B-E-S (fig. 5).* The meadow has vege-

Figure 4—Meadow series vegetated margin (A), subalpine (A), basin (A), lotic (D), and beaked sedge (H) in Sky Parlor Meadow, Sequoia National Park.

Figure 5—Meadow series vegetated margin(A), montane (B), slope (B), xeric (E), and bluegrass (S) in Crane Flat Meadow, Yosemite National Park.

Figure 6—Meadow series vegetated margin (A), subalpine (A), basin (A), normal (C), and shorthair (C) in Delany Meadow, Yosemite National Park.

tated margins and lies at 6,430 ft (1,960 m) in the montane belt. It is a slope meadow. The site is somewhat elevated above the main meadow area and has a 3.5 percent slope. Hydrologically, the site must be considered in the xeric series. The soil is well drained with a sandy loam texture. It receives no moisture other than normal precipitation. The water table, if present, appears quite deep and likely has little influence on the site.

Floristically, the site is in the bluegrass *(Poa)* series. More than 60 percent of the composition is Kentucky bluegrass *(Poa pratensis).* A forb, dusky horkelia *(Horkelia fusca),* is the next most frequent species.

Soil organic matter is about 10 percent and pH is 5.5. Litter covers 74 percent and bare soil, 21 percent of the surface. Gopher mounds are abundant on the site. This situation occurs on many xeric sites, especially where dryness has resulted from a lowered water table due to erosion.

The site described here shows no evidence of recent erosion, and it has been ungrazed for many years. Under grazing, some reduction in bluegrass may occur, and other species of forbs will become more abundant.

Series A-A-A-C-C is represented by a site *(fig. 6)* one-half mile North of Dog Lake in Yosemite National Park. The meadow has vegetated margins, lies at 9,400 ft (2,865 m) in the

Figure 7—Meadow series vegetated margin (A), montane (B), slope (B), hanging (B), and tinkers penny (N) in McKinley Grove, Kings River Ranger District, Sierra National Forest.

Figure 8—Meadow series vegetated margin (A), montane (B), basin (A), sunken-concave (F), and slender spikerush (L) near Clover Meadow, Minarets Ranger District, Sierra National Forest.

subalpine belt, and is in a basin. An intermittent stream meanders the length and is tributary to Delaney Creek, which passes through the lower reaches of the meadow.

The hydrologic series of the site is normal. It receives water from upslope in the spring. Soil texture at the site is a sandy loam with pH 4.9. The water table in mid-July was at a depth of 78 cm. At that depth, a layer of coarse sand appears to serve as an aquifer carrying water under the meadow. Shorthair is the vegetative series of the site. Shorthair and Sierra ricegrass *(Oryzopsis kingii)* make up 36 percent of the composition. On this site, however, Sierra ricegrass is the main species. On the drier side, sites of this series tend to merge with sites having short-hair sedge. On the more moist side, sites of this series tend to have more shorthair. The caespitose grasses form large bunches, which account for much of the 12 percent cover of higher plants. Litter covers 77 percent of the surface. Alpine aster *(Aster alpigenus)* and Sierra bilberry are the two next most important species.

A meadow in the McKinley Grove of Sequoias on the Kings River Ranger District, Sierra National Forest, contains a fine example of *series A-B-B-B-N (fig. 7).* The meadow has vegetated margins, lies at 6,480 ft (1,975 m) in the montane belt, and has a 15 percent slope.

Hydrologically, the site is in the hanging series. Seeps

Figure 9—Meadow series vegetated margin (A), subalpine (A), slope (B), xeric (E), and short-hair sedge (E) in Dana Meadow, Yosemite National Park.

provide a constant supply of water for the site and the meadow as a whole. Water appears to be retained near the surface by a slowly permeable zone in the 10- to 20-cm soil layer. Soil texture is a sandy loam with pH 5.2 and 29 percent organic matter. Below about 20 cm, the amount of sand and gravel increase. The dominant surface characteristic, with 50 percent cover, is moss. Floristically, the site is in the tinkers penny *(Hypericum anagalloides)* series—the hillside bog class of Ratliff (1979, 1982). Sites of this series vary markedly in species composition. Tinkers penny, American bistort *(Polygonum bistortoides)*, and violet (*Viola* spp.) should, however, be found. Here, tinkers penny and Pacific cowbane *(Oxypolis occidentalis)* together comprise 38 percent of the nearest shoot-to-point composition.

An ephemeral-lake site *(fig. 8)* on the Minarets Ranger District, Sierra National Forest, represents *series A-B-A-F-L*. The meadow has vegetated margins, is in the montane belt at 7,140 ft (2,176 m), and is a basin type. Hydrologically, the site is classed as sunken-concave, and water may stand on it until mid-summer. By fall, the water table may be at a considerable depth, and the surface soil may be dry and hard.

Slight changes in normal water depth alter species composition from point to point in an ephemeral-lake. The slender spikerush vegetative series is typical of ephemeral-lake sites. Here slender spikerush and creeping spikerush *(Heleocharis palustris)* are the predominant species. Other species characteristic of the class include clover fern *(Marsilea vestita)* and *Porterella carnosula*.

Bare soil on this site occupies 43 percent of the surface and litter occupies 54 percent of the surface. Soil pH is 6.0 and soil texture is a silty clay loam.

Dana Meadow in Yosemite National Park contains several examples of *series A-A-B-E-E*. The meadow has vegetated margins, is in the subalpine belt, and is slope type in its upper reaches. The representative site *(fig. 9)* lies at 9,860 ft (3,005 m) and occupies a ridge with a general slope of 5 percent. The soil is a sandy loam with pH 5.1 and 7 percent organic matter. Hydrologically, the site is xeric—the expected hydrologic series for sites of the short-hair sedge vegetative series. Short-hair sedge makes up 80 percent of the composition, and its patches of dense sod largely account for the high plant cover. Several species, including Brewer's lupine *(Lupinus breweri)*, Heretic penstemon *(Penstemon hetrodoxus)*, and skyline bluegrass *(Poa epilis)*, occupy the spaces between sod patches.

Meadows with sandy margins in the subalpine belt at an elevation of 10,800 ft (3,293 m) are represented in Sequoia National Park at Siberian Outpost *(fig. 10)*. Specific data on sandy margin meadows is not available. Their occurrence appears to be correlated with the southern limits of glaciation (Benedict and Major 1982). Siberian Outpost contains basin, stream, and slope meadow sites. A few hanging and lotic sites are found, but most are normal or xeric. Floristically, the main series are shorthair, beaked sedge, fewflowered spikerush, and short-hair sedge. The sandy areas contain the *Eriogonum-Oreonana clementis* association (Benedict 1981) of the buckwheat *(Eriogonum)* series. In some places, western needlegrass *(Stipa occidentalis)* and bottlebrush squirreltail *(Sitanion hystrix)* are abundant.

PRODUCTIVITY OF MEADOWS

Productivity of meadows is related to site elevation, vegetation type, range condition, fertilization, and degree of utilization. Productivity of meadows generally decreases as elevation increases. Vegetation of mesic meadow sites tends to be the most productive, but it may also be the most variable from

Figure 10—Siberian Outpost, Sequoia National Park, with sandy margin (B) meadows.

year to year. As range conditions decline, meadow productivity generally declines. Productivity usually peaks when vegetation is at or near climax. Nitrogen fertilization at rates of 90 to 150 lb N per acre (100 to 168 kg/ha) and phosphorus fertilization at 40 to 80 lb P_2O_5 per acre (45 to 90 kg/ha) frequently increase production many times over that of unfertilized meadows. Responses to phosphorus have been more consistent than responses to nitrogen. When used to reduce soil acidity, lime may increase meadow productivity. With fertilization, species composition may change and the resulting increases in productivity will be temporary.

Elevation

Productivity of meadows in excellent condition decreased as elevation increased (Crane 1950). Cow months (a month's tenure by one cow) per acre declined from 3.4 at 5,500 to 6,000 ft (1,676 to 1,829 m) to 1.7 at elevations of 8,500 to 9,000 ft (2,591 to 2,743 m). Similar trends were found along Rock Creek in Sequoia National Park (Giffen and others 1970). At 9,450 ft (2,987 m) elevation meadow productivity across vegetation types was about 1,695 lb per acre (1,900 kg/ha). At 11,600 ft (3,536 m) productivity was only 312 lb per acre (350 kg/ha).

On the Sierra National Forest, elevation of Markwood Meadow is 5,800 ft (1,768 m) and of Exchequer Meadow 7,280 ft (2,219 m). At Markwood Meadow, forage production was estimated at 3,739 to 4,508 lb per acre (4,191 to 5,053 kg/ha) (Clayton 1974, Pattee 1973). At Exchequer Meadow, forage production was estimated at 1,280 to 2,963 lb per acre (1,435 to 3,221 kg/ha).

The general trend to lower productivity with increased elevation should be considered in arriving at grazing capacities and in considering meadow condition and trend. Low production at high elevations does not indicate poor condition, nor does high production at low elevations indicate good condition.

Vegetation

Series

Data on meadow productivity are of more value when they are related to vegetative series. Such data are scant and available for only a few series. Sources of the data provided *(table 7)* include Sanderson (1967) and Ratliff (1974). Data for the slender spikerush series comes from similar sites on the Lassen National Forest (Reppert and Ratliff 1968). Other data were collected from 1971 to 1981 in the Sequoia and Kings Canyon National Parks and the Sierra National Forest.

Maximum productivity is achieved by those vegetative series that occur on more mesic sites (Giffen and others 1970). Slender spikerush and beaked sedge sites generally are wetter than sites with Nebraska sedge and tufted hairgrass. Sites with short-hair sedge are the most xeric. The effects of elevation are also evident *(table 7);* the last three series generally are found at higher elevations than the others.

Table 7—*Estimates of herbage production for seven meadow vegetative series of the Sierra Nevada, California*

Vegetative series	Sites studied	Herbage production		
		gm/m²	*kg/ha*	*lb/acre*
Beaked sedge	7	185	1,850	1,650
Slender spikerush	2	113	1,130	1,010
Tufted hairgrass	5	270	2,695	2,405
Nebraska sedge	3	314	3,145	2,805
Fewflowered spikerush	1	128	1,280	1,145
Shorthair	28	119	1,195	1,065
Short-hair sedge	11	32	325	285

Table 8—*September standing crops at five meadow sites in Sequoia National Park, California*

Site	Vegetative series	Standing crops 1972	1973	1974
			gm/m²	
Hair Sedge	Short-hair sedge	45.2a[1]	33.8a	50.6a
Lake Shore	Shorthair	199.8a	182.2a	296.8a
Chagoopa	Shorthair	132.0b	164.2b	274.8a
Big Arroyo	Tufted hairgrass	467.3a	378.6ab	303.2b
Sky Parlor	Beaked sedge	227.5a	195.0a	260.5a

[1]Values, within rows, followed by the same letter do not differ significantly (P = 0.95) by Tukey's w-procedure.

Data on yearly variations in productivity are available from two studies in Sequoia National Park (Ratliff 1976, 1980). Among the five sites in these studies, standing crops differed significantly between years at two: Chagoopa and Big Arroyo *(table 8)*. These two sites are the more mesic of the five. The results suggest that year-to-year variations in productivity may be greater on mesic than on drier or wetter meadow sites.

Standing crops at four of these five meadow sites approximated the maximum values found elsewhere in Sequoia National Park. The values *(table 7)* for the beaked sedge, fewflowered spikerush, shorthair, and short-hair sedge vegetative series fall within the ranges DeBenedetti (1983) found elsewhere in the park *(table 9)*. It is acceptable to assume, therefore, that productivity on sites of those series will usually fall within those values. But the differences among the values from different sources show that it is unacceptable to assume a standard productivity for a given vegetative series.

Species

Estimated production per unit of composition times the observed composition provides an estimate of productivity of a particular species on a site. The sum of the species productivities estimates site productivity. This type of information is expensive to obtain, however.

The only published information on individual species production for meadows of the Sierra Nevada is given by Sanderson (1967). He reported average production from two meadows for Nebraska sedge (123 lb per acre or 138 kg/ha), pullup muhly (239 lb per acre or 268 kg/ha), California oatgrass *(Danthonia californica americana)* (24 lb per acre or 27 kg/ha), and Idaho bentgrass *(Agrostis idahoensis)* (6 lb per acre or 7 kg/ha). Legumes as a group produced 30 lb per acre (34 kg/ha). Grasslike plants as a group (other than Nebraska sedge) produced 582 lb per acre (652 kg/ha).

In meadows of the Sierra Nevada, I have observed more than 200 herbaceous and shrubby plant species. But the listings provided in this section do not exhaust the possibilities *(table 10)*. Each new meadow site should be considered a resource of new species.

Plant species can be categorized as decreasers, increasers, and invaders (Bell 1973, Dyksterhuis 1949, Range Term Glossary Committee 1974, Stoddart and others 1975). Categorization of the species in this report is based on my experience and on information in the literature (Dayton 1960; Hayes and Garrison 1960; Hermann 1966, 1970, 1975; Hitchcock 1950; Munz and Keck 1959; U.S. Dep. Agric., Forest Serv. 1937; Weeden 1981). Decreaser species, present in climax vegetation, decrease in the stand with overgrazing. Increaser species, present in climax vegetation, increase in the stand—at least initially—with overgrazing. Invader species, absent in climax vegetation, increase in the stand with overgrazing.

Decreasers and Increasers—Decreaser species are usually major constituents of the composition at climax. Increaser species are usually minor constituents of the composition at climax. Under certain conditions, however, some such species are probably true decreasers—*Habenaria dilatata* (white bogorchid), for example. Also, some increaser species indicate a greater departure from climax than others—*Cirsium drummondii* (dwarf thistle) compared with *Danthonia intermedia* (timber oatgrass), for example. Species believed to represent both situations are indicated in the tables.

The presence or even abundance of various species may or may not indicate overgrazing (Sharsmith 1959). The lists, therefore, should be used only as guides. The land manager must decide, on a site-by-site basis, which species are the decreasers, increasers, and invaders.

Carbohydrate Storage—Carbohydrate storage cycles in meadow species need to be considered in determining proper use. Carbohydrate storage patterns are probably related to a particular species' pattern of growth and reproduction (Hyder and Sneva 1959). The responses of one species cannot, therefore, always serve to explain those of another species (Smith 1972).

The level of carbohydrate reserves in plants is affected by rates of photosynthesis, respiration, and growth (Cook 1966). Carbohydrate reserves in grasses gradually decline during winter because of continued, though slight, respiration. A sharp decline usually occurs at the start of spring growth. And reserves build up during maturation (Heady 1975). Defoliation will cause a temporary drop in carbohydrate reserves.

I know of only one study specifically on carbohydrate storage cycles of meadow plant species in the Sierra Nevada (Steele 1981, Steele and others 1984). Seasonal variation in total nonstructural carbohydrate (TNC) levels in rhizomes

Table 9—*Ranges in herbage standing crops and average standing crops at quiescence for five vegetative series in Sequoia National Park, 1977-1981*

Vegetative series	5-Year range[1]	Average standing crop		
	gm/m²	*gm/m²*	*kg/ha*	*lb/acre*
Beaked sedge	55 to 367	167	1668	1488
Fewflowered spikerush (≤ 8 cm tall)	12 to 97	53	532	475
Fewflowered spikerush (> 8 cm tall)	113 to 415	238	2379	2122
Shorthair	18 to 292	74	742	662
Short-hair sedge	6 to 46	22	222	198
Brewer sedge[2]	15 to 68	29	288	257

Source: DeBenedetti (1983)

[1]Low- and high-plot values observed during study.

[2]*Carex breweri.* Not listed in *table 2*. Hydrologically, sites of the series are xeric.

Table 10—*Ecological groups of meadow species in the Sierra Nevada, California*

Code[1]	Scientific name[2]	Common name	Plant belt[3]
	Decreaser or primary meadow species		
AGSU	*Agropyron subsecundum*	Bearded slender wheatgrass	M
AGDI	*Agrostis diegoensis*	Thin bent	M
AGEX-1	*Agrostis exarata*	Spike bent	M
ALL-2	*Allium* sp. herbaceous perennial	Wild onion	M-S
ALAE	*Alopecurus aequalis*	Shortawn foxtail	M
BOMU	*Botrychium multifidum*	Broadleaf grape-fern	M
BOSI	*Botrychium simplex*	Little grape-fern	S
CABR-1	*Calamagrostis breweri*	Shorthair	S-A
CACA-1	*Calamagrostis canadensis*	Bluejoint reedgrass	M-S
CAAB-2	*Carex abrupta*	Abruptbeak sedge	M-S-A
CAAQ	*Carex aquatilis*	Sand sedge	M-S
CAAT-2	*Carex athrostachya*	Slenderbeak sedge	M
CAEX-1	*Carex exserta*	Short-hair sedge	S
CAFI-2	*Carex fissuricola*	—	M-S
CAFR-1	*Carex fracta*	—	M-S
CAHA-3	*Carex hassei*	—	M
CAHE-3	*Carex heteroneura*	—	M-S-A
CAJO	*Carex jonesii*	Jones sedge	M-S
CALA-3	*Carex lanuginosa*	Wooly sedge	M-S
CALE-2	*Carex lemmonii*	—	M-S
CAMI-1	*Carex microptera*	Small-wing sedge	M-S
CAMU-2	*Carex multicostata*	Manyrib sedge	M-S
CANE-1	*Carex nebraskensis*	Nebraska sedge	M-S
CANE-2	*Carex nervina*	—	M-S
CANI	*Carex nigricans*	Black alpine sedge	S-A
CAOR	*Carex ormantha*	Star of David sedge	M-S
CARO-2	*Carex rostrata*	Beaked sedge	M-S
CASC-2	*Carex scopulorum*	—	M-S
CASE-1	*Carex senta*	Rough sedge	M
CASI-1	*Carex simulata*	Analogne sedge	M-S
CASP-2	*Carex spectabilis*	Showy sedge	M-S-A
CASU-3	*Carex subnigricans*	—	S-A
CAVE-2	*Carex vesicaria*	Blister sedge	M-S
CACU-2	*Castilleja culbertsonii*	Indian paint-brush	S-A
DECA-1	*Deschampsia caespitosa*	Tufted hairgrass	M-S
GLEL	*Glyceria elata*	Tall mannagrass	M
GLLE-1	*Glyceria leptostachya*	Slimleaf mannagrass	M
GLST	*Glyceria striata*	Fowl mannagrass	M
HADI	*Habenaria dilatata*	White bogorchid	M
HEPA-2	*Heleocharis palustris*	Creeping spikerush	M
JUBA	*Juncus balticus*	Baltic rush	M
JUEN	*Juncus ensifolius*	—	M
JUME-2	*Juncus mertensianus*	Merten's rush	M
JUME-3	*Juncus mexicanus*	Twisted baltic rush	M-S
JUNE	*Juncus nevadensis*	Nevada rush	M
JUOR	*Juncus orthophyllus*	Straightleaf rush	M-S
JUOX	*Juncus oxymeris*	Pointed rush	M
LUCO-1	*Luzula comosa*	Maryflowered wood-rush	M

Table 10—*Ecological groups of meadow species in the Sierra Nevada, California (continued)*

Code[1]	Scientific name[2]	Common name	Plant belt[3]
	Decreaser or primary meadow species (continued)		
LUOR	*Luzula orerstera*	Common wood-rush	S-A
MAVE	*Marsilea vestita*	Clover fern	M
MOCH	*Montia chamissoi*	Chamisso miner's lettuce	M-S-A
ORKI	*Oryzopsis kingii*	Sierra ricegrass	S
OXOC	*Oxypolis occidentalis*	Pacific cowbane	M
PEPRF	*Penstemon procerus formosus*	Small-flowered penstemon	M-S
PEBO	*Perideridia bolanderi*	Bolander yampah	M-S
PEPA-5	*Perideridia parishii*	Parish's yampah	M-S
PHAL-1	*Phleum alpinum*	Alpine timothy	S-A
POAMS	*Polygonum amphibium stipulaceum*	Ladys thumb knotweed	M
SAL-11	*Salix* sp.	Willow	M-S-A
SIPR	*Sibbaldia procumbens*	—	S-A
SIRE	*Sidalcea reptans*	Spike checker-mallow	M
SIEL	*Sisyrinchium elmeri*	Elmer's yellow-eyed grass	M
SPRO	*Spiranthes romanzoffiana*	Hooded ladies tresses	M
TRWO-2	*Trifolium wormskioldii*	Cow clover	M
TRSP-1	*Trisetum spicatum*	Spike trisetum	S-A
TRWO-1	*Trisetum wolfii*	Beardless trisetum	M-S
VANI	*Vaccinium nivictum*	Sierra bilberry	S-A
	Increaser or secondary meadow species		
ACLA-2	*Achillea lanulosa*	Western yarrow	M-S-A
AGTR-1	*Agropyron trachycaulum*	Slender wheatgrass	M
AGID	*Agrostis idahoensis*	Idaho bent	M
AGLE	*Agrostis lepida*	Sequoia bent	M-S
AGSC-2	*Agrostis scabra*	Rough bent	M
AGVA	*Agrostis variabilis*	Mountain bent	S-A
ANCO-1	*Antennaria corymbosa*	Flattop pussytoes	M-S
ANRO	*Antennaria rosea*	Rose pussytoes	M-S-A
ASAD	*Aster adscendens*	Long-leaf aster	M
ASALA	*Aster alpigenus andersonii*	Alpine aster	S-A
ASOC	*Aster occidentalis*	Western aster	M
ATFIC	*Athyrium filix-femina californicum*	Common lady-fern*	M
CAMI-2	*Calochortus minimus*	Lesser star tulip	M
CAHO-2	*Caltha howellii*	Twinflower marshmarigold*	M
CACA-3	*Carex canescens*	Silvery sedge	M-S
CADO	*Carex douglasii*	Douglas sedge	M-S-A
CAFE-2	*Carex feta*	Western sedge	M
CAIN-3	*Carex integra*	Smoothbeak sedge	M-S
CALE-5	*Carex leptopoda*	Short-scale sedge	M
CALU-1	*Carex luzulaefolia*	—	M
CIDR	*Cirsium drummondii*	Dwarf thistle	M
CITI	*Cirsium tioganum*	Tioga thistle	S
DACA-1	*Danthonia californica americana*	California oatgrass	M
DAIN	*Danthonia intermedia*	Timber oatgrass	M-S
DOAL	*Dodecatheon alpinum*	Alpine shooting-star	S-A
DOJE	*Dodecatheon jeffreyi*	Jeffrey shooting-star	M

Table 10—*Ecological groups of meadow species in the Sierra Nevada, California (continued)*

Code[1]	Scientific name[2]	Common name	Plant belt[3]
	Increaser or secondary meadow species (continued)		
DORO	*Drosera rotundifolia*	Roundleaf sundew	M
ELGL	*Elymus glaucus*	Blue wildrye	M
EPBR	*Epilobium brevistylum*	Sierra willow-herb	M
EPGL-1	*Epilobium glaberrimum*	Glaucous willow-herb	M-S-A
EPOR	*Epilobium oregonense*	Oregon willow-herb	M-S-A
EQAR	*Equisetum arvense*	Field horsetail*	M
ERCO-5	*Erigeron coulteri*	Coulter fleabane	M-S
ERPEH	*Erigeron peregrinus hirsutus*	Peregrine fleabane	M-S
FRPL-1	*Fragaria platypetala*	Broad petal strawberry	M
GATRS	*Galium trifidum subbiflorum*	Sweet scented bedstraw	M
GEAM	*Gentiana amarella*	Annual gentian	M-S
GEHO	*Gentiana holopetala*	Tufted gentian	M-S
GENE-1	*Gentiana newberryi*	Newberry gentian	S-A
GESI	*Gentiana simplex*	Hikers gentian	M-S
GNPA	*Gnaphalium palustre*	Lowland cudweed*	M
HAAP	*Haplopappus apargioides*	Alpine flames	S-A
HEBI	*Helenium bigelovii*	Bigelow sneezeweed*	M
HEHO	*Helenium hoopesii*	Orange sneezeweed*	M
HEAC	*Heleocharis acicularis*	Slender spikerush	M
HEPA-4	*Heleocharis pauciflora*	Fewflowered spikerush	M-S-A
HEPU-3	*Hesperochiron pumilus*	Meadow centaur	M
HOBR	*Hordeum brachyantherum*	Meadow barley	M
HYAN	*Hypericum anagalloides*	Tinkers penny	M
HYFOS	*Hypericum formosum*	Southwestern St. Johnwort	M
IVLY	*Ivesia lycopodioides*	Club moss ivesia	S
IVPU	*Ivesia purpurascens*	Purple ivesia	M
JUDR	*Juncus drummondii*	Drummond rush	M-S
JUPA-1	*Juncus parryi*	Parry rush	S-A
KAPOM	*Kalmia polifolia microphylla*	Alpine laurel*	S-A
LENE-2	*Lewisia nevadensis*	Nevada lewisia	M-S
LOOB	*Lotus oblongifolius*	Stream deervetch	M
LUPR	*Lupinus pratensis*	Inyo meadow lupine*	M-S
MECIS	*Mertensia ciliata*	Mountain bluebells	M
MIMO-3	*Mimulus moschatus*	Musk monkeyflower	M
MIPR	*Mimulus primuloides*	Primrose monkeyflower	M-S
MITI	*Mimulus tilingii*	Mountain monkeyflower	M-S-A
MUFI	*Muhlenbergia filiformis*	Pullup muhly	M-S
MURI-1	*Muhlenbergia richardsonis*	Mat muhly	M-S
NOAL	*Nothocalais alpestris*	Alpine lake-agoseris	M-S-A
OESA	*Oenanthe sarmentosa*	Pacific water-drop-wort	M
PEAT	*Pedicularis attolens*	Little elephant heads	M-S
PEGR-1	*Pedicularis groenlandica*	Elephant heads	M-S
PEHE-2	*Penstemon hetrodoxus*	Heretic penstemon	S-A
PEOR-1	*Penstemon oreocharis*	—	M
PHBO-2	*Phalacroseris bolanderi*	Bolander's sunflower	M
PHBR-4	*Phyllodoce breweri*	Mountain-heather*	S-A
POEP-1	*Poa epilis*	Skyline bluegrass	S-A
POPR-1	*Poa pratensis*	Kentucky bluegrass	M

Table 10—*Ecological groups of meadow species in the Sierra Nevada, California (continued)*

Code[1]	Scientific name[2]	Common name	Plant belt[3]
	Increaser or secondary meadow species (continued)		
POBI-3	*Polygonum bistortoides*	American bistort	M-S
POCA-7	*Porterella carnosula*	—	M-S
POBR	*Potentilla breweri*	Brewer cinquefoil	M-S-A
POFL-1	*Potentilla flabellifolia*	Fanleaf cinquefoil	M-S-A
POGL-2	*Potentilla glandulosa*	Sticky cinquefoil	M
POGR-4	*Potentilla gracilis nuttallii*	Fivefinger cinquefoil	M
PUER	*Puccinellia erecta*	Upright alkali grass	S-A
PUPA-2	*Puccinellia pauciflora*	Fewflowered alkali grass	M
RAAL-1	*Ranunculus alismaefolius*	Plantainleaf buttercup	M
RAFLO	*Ranunculus flammula ovalis*	Spearwort buttercup*	M
RHOC	*Rhododendron occidentale*	Western azalea*	M
ROCU	*Rorippa curvisiliqua*	Western yellow cress	M
RUCA-1	*Rudbeckia californica*	California coneflower	M
SASAH	*Sagina saginoides hesperia*	Arctic pearlwort	M-S-A
SAAP-2	*Saxifraga aprica*	Sierra saxifrage	M-S-A
SAOR-1	*Saxifraga oregana*	Oregon saxifrage	M
SAPUA	*Saxifraga punctata arguta*	Dotted saxifrage	M-S
SCCL	*Scirpus clementis*	Slender club-rush	S-A
SCCO	*Scirpus congdoni*	Congdon's bulrush	M
SCCR	*Scirpus criniger*	Fringed bulrush	M-S
SCMI	*Scirpus microcarpus*	Panicled bulrush	M
SECL-1	*Senecio clarkianus*	—*	M
SEINM	*Senecio integerrimus major*	Lambstongue groundsel*	M
SOCAE	*Solidago canadensis*	Canada goldenrod	M
SOMU	*Solidago multiradiata*	Alpine goldenrod	S-A
STLO-1	*Stellaria longipes*	Long-stalked starwort	M
STCO-1	*Stipa columbiana*	Subalpine needlegrass	M-S
THA-1	*Thalictrum* sp.	Meadow rue	M-S
TOGLO	*Tofieldia glutinosa occidentalis*	Western tofieldia*	M-S
TRBO	*Trifolium bolanderi*	Bolanders clover	M
TRLO	*Trifolium longipes*	Longstalk clover	M
TRMO-1	*Trifolium monanthum*	Carpet clover	M
TRCEP	*Trisetum cernuum projectum*	Nodding trisetum	M
VECA-1	*Veratrum californicum*	Western false-hellebore*	M
VEALA	*Veronica alpina alterniflora*	Alpine speedwell	M-S
VEAM	*Veronica americana*	American speedwell	M
VESC-2	*Veronica scutellata*	Marsh speedwell	M
VIMA-2	*Viola macloskeyi*	Western sweet white violet	M-S
	Invader or low-value meadow species		
ACMI	*Achillea millefolium*	Common yarrow	M
ANDI	*Antennaria dimorpha*	Low pussytoes	M-S
BRLA-2	*Brodiaea laxa*	Grassnut brodiea	M
BRTE	*Bromus tectorum*	Cheatgrass brome	M
CAUM-2	*Calyptridium umbellatum*	Pussy paws*	M-S
CARO-1	*Carex rossii*	Ross sedge	M-S
CIALP	*Circaea alpina pacifica*	Enchanter's nightshade	M
CIAN	*Circium andersonii*	Anderson's thistle	M

Table 10—*Ecological groups of meadow species in the Sierra Nevada, California (continued)*

Code[1]	Scientific name[2]	Common name	Plant belt[3]
	Invader or low-value meadow species (continued)		
DAGL-1	*Dactylis glomerata*	Orchardgrass	M
DECA-4	*Dentaria californica*	California toothwort	M
DEDA	*Deschampsia danthonoides*	Annual hairgrass	M
FEOC-1	*Festuca occidentalis*	Western fescue	M
FRCA-1	*Fragaria californica*	California strawberry	M
GANU-1	*Gayophytum nuttallii*	Nuttal groundsmoke	M
HIAL	*Hieracium albiflorum*	White hawkweed	M
HOFUC	*Horkelia fusca capitata*	Dusky horkelia	M
HOFUP1	*Horkelia fusca parviflora*	—	M-S
HOLA-1	*Holcus lanatus*	Velvet grass	M
IRHA	*Iris hartwegii*	Foothill iris	M
IVUN	*Ivesia unguiculata*	Yosemite ivesia	M
JUBU	*Juncus bufonius*	Toad rush	M
JUTE-1	*Juncus tenuis*	Poverty rush	M
JUTEC	*Juncus tenuis congestus*	—	M
LETR-2	*Lewisia triphylla*	Three-leaf lewisia	M-S
LONE-3	*Lotus nevadensis*	Nevada deervetch	M
LOPU-2	*Lotus purshianus*	Pursh deervetch	M
LUBRB	*Lupinus breweri bryoides*	Brewer's lupine*	S-A
LUCO-5	*Lupinus covillei*	Coville lupine*	M-S
MAGL-1	*Madia glomerata*	Cluster tarweed	M
MIPE	*Mitella pentandra*	Fivestamen miterwort	M
MUMI-2	*Muhlenbergia minutissima*	Annual muhly	M
NESP	*Nemophila spatulata*	Sierra nemophila	M
PHPR-1	*Phleum pratense*	Timothy	M
PHDI-4	*Phlox diffusa*	Spreading phlox	M-S-A
PLHI-1	*Plagiobothrys hispidulus*	Hairy popcorn flower	M
PODO-3	*Polygonum douglasii*	Douglas knotweed	M
POKE-2	*Polygonum kellogii*	Kellogg knotweed	M-S-A
PRVU	*Prunella vulgaris*	Common self-heal	M
RORA	*Rotala ramosior*	Common toothcup	M
RUAN	*Rumex angiocarpus*	Sheep sorrel	M
SIHY	*Sitanion hystrix*	Bottlebrush squirreltail	M-S
STAL	*Stachys albens*	Whitestem hedge nettle	M-S
STOC-1	*Stipa occidentalis*	Western needlegrass	M-S
TAOF	*Taraxacum officinale*	Common dandelion	M
TRMI-2	*Trifolium microcephalum*	Littlehead clover	M
TRRE	*Trifolium repens*	White clover	M
VESE	*Veronica serpyllifolia*	Thymeleaf speedwell	M
VIAD	*Viola adunca*	Hookedspur violet	M

[1]Codes from Reed and others (1963). Decreaser species whose codes are italicized are expected to comprise a relatively small percentage of the composition at climax, but they are expected to decrease with overgrazing. When in abundance, increaser species whose codes are italicized may indicate a greater degree of deterioration from climax than other increaser species.

[2]Scientific names from Munz and Keck (1959).

[3]M = montane; S = subalpine; A = alpine.

*These species are known to be or are suspected of being poisonous to some classes of livestock.

and shoots of a natural population of Nebraska sedge were studied at Tule Meadow on the Sierra National Forest. The TNC cycle for Nebraska sedge follows closely the generalized cycle characterized earlier (Humphreys 1966). TNC in rhizomes decreased to 7.5 percent of the dry weight during early shoot growth, and reached a peak level of 17.4 percent in fall. TNC levels in shoots ranged from a low of 10.6 percent in spring to a high of 16 percent in late summer after flowering. TNC levels in emerging shoots average 16.4 percent in September and 19.1 percent at the end of October. Carbohydrates appear to have been transferred from mature shoots to emerging shoots serving as storage locations or sinks.

In Sequoia National Park, reductions of 20 percent to 40 percent in reserves of TNC in roots of short-hair sedge, shorthair, and Carex *scopulorum* one year after clipping the herbage to a one-inch (2.5-cm) stubble height were reported (DeBenedetti 1980). And near Carson Pass, decreases in the carbohydrate content of roots and rhizomes of American bistort, *Sibbaldia procumbens*, and twinflower marshmarigold *(Caltha howellii)* when plants were transplanted from 9,000 to 6,000 ft (2,743 to 1,829 m) elevation were reported (Mooney and Billings 1965). Naturally growing pussy paws *(Calyptridium umbellatum)*, however, had about 50 percent more carbohydrate at the lower than higher elevation.

Naturally growing American bistort in Wyoming showed the same general response to elevation as did pussy paws at Carson Pass (Mooney and Billings 1965). The reduced carbohydrate levels of the transplanted plants were attributed to higher respiration rates at the lower, warmer elevations. Naturally growing plants of high-elevation species at lower elevations apparently compensate for the higher respiration by being able to store relatively high amounts of carbohydrates.

Underground organs of herbaceous alpine species in the Medicine Bow Mountains, Wyoming, contained abundant reserves of carbohydrates (Mooney and Billings 1960). Many of the species apparently use much of their reserves for growth before and immediately after snowmelt. Because of rapid development, it appears that the reserves are restored quickly and maintained during most of the growing season. Shoot and root reserves, however, may experience a depression associated with the period of flowing and fruit set. Carbohydrates in rhizomes of American bisort declined sharply from when leaves were expanding to when plants were in bud. The reserves then rapidly rebuilt to early bloom followed by a lower rate of buildup to dormancy. These findings suggest that American bistort could be most easily damaged by defoliation when it is in bud. Each species studied throughout the season showed somewhat different trends in its carbohydrate cycle. Mooney and Billings consider that slower developing species (such as species of *Gentiana*) may depend more upon current photosynthesis than upon a reserve of carbohydrates.

Rhizome reserves of *Carex bigelowii* on Mt. Washington, New Hampshire, were 61 percent depleted during initial growth (Fonda and Bliss 1966). The reserves were fully recovered during flowering but dropped by about 30 percent immediately afterwards. By seed dispersal, however, the reserves were again fully recovered. Although the rhizomes of grasslikes are important, roots of grasslike species are considered to be relatively unimportant as storage organs. A species of cinquefoil *(Potentilla)* and a species of huckleberry *(Vaccinium)* both showed lower carbohydrate levels after full bloom than before flowering.

Maximum carbohydrate storage in timothy *(Phleum pratense)* occurred between anthesis and the seed-in-dough stage (Reynolds and Smith 1962). With smooth bromegrass *(Bromus inermis)*, a drop in carbohydrate storage occurred as the heads emerged from the boot, after which carbohydrates increased to maturity. With both species of grass and with alfalfa *(Medicago sativa)*, the peaks were followed by slight drops, but total available carbohydrates (TAC) generally remained high to the end of the season.

Timothy is often cut for hay. Carbohydrate trends in timothy and other "northern grasses" were studied (Okajima and Smith 1964). For timothy, total available carbohydrates decreased with growth of new shoots in early spring and were minimum when tillers started to elongate. Carbohydrates then accumulated and were high at heading when timothy was cut. It was cut twice during the season, and with each cutting the general cycle of TAC depletion and storage was repeated.

Higher levels of carbohydrates in timothy were obtained with two cuttings than with three cuttings (Reynolds and Smith 1962). And timothy stored higher percentages of carbohydrates with all cutting treatments than either alfalfa or smooth bromegrass.

Two groups of perennial grasses can be distinguished by the type of carbohydrates stored in overwintering organs (Okajima and Smith 1964). One group accumulates fructosan and sucrose; the other accumulates starch and sucrose. The first group contains grasses native to temperate latitudes. The second group contains grasses native to semitropical or tropical latitudes. Timothy, smooth bromegrass, orchardgrass *(Dactylis glomerata)*, reed canarygrass *(Phalaris arundinacea)*, and Kentucky bluegrass were all found to store fructosan. Smooth bromegrass had, however, mostly sucrose. *Agrostis alba*, *A. tenuis*, and *Bromus marginatus* also store fructosan (Ojima and Isawa 1968).

These results suggest that most meadow grasses in the Sierra Nevada store fructosan. Whether this is true for the grasslikes is not known. For clovers (*Trifolium* spp.), the main carbohydrate reserve is starch (Ojima and Isawa 1968). American bistort apparently stores starch, also (Mooney and Billings 1960). At least some meadow forbs, therefore, appear related to semitropical or tropical grasses in the type of carbohydrates stored.

Among other species of range plants, the carbohydrate reserves of lambstongue groundsel *(Senecio integerrimus)* were studied (Donart 1969). This and other species of the genus grow in meadows of the Sierra Nevada. When the plants were in full leaf—just prior to formation of flower buds, the root reserves were at their seasonal maximum. When the flower buds began to develop, the reserves dropped 69 percent. With flowering, the reserves rose to about one-half the maximum, declined slightly in the late flowering stage, and then increased to the end of the growing season. The plants were generally more advanced phenologically at given dates than other species studied (Donart 1969). Management of *Senecio* species may require therefore a different treatment than grasses and various other forbs.

The studies on carbohydrate cycles and reserves suggest that grazing is detrimental when reserves are being spent to produce spring growth or near the time of heading. I suggest that the range readiness standards (U.S. Dep. Agric., Forest Serv. 1969) are adequate for montane meadows. The range is considered ready for grazing when Kentucky bluegrass, Nebraska sedge, and/or tufted hairgrass growth is 6 inches (15 cm) tall. By then, carbohydrate reserves should be sufficient for plants to withstand moderate defoliation.

A late grazing that prevents carbohydrate accumulation in emerging shoots could be damaging. I suggest that grazing should therefore cease in fall in time to permit regrowth sufficient to store carbohydrates for winter respiration and initial spring growth.

Accumulation of carbohydrate reserves in plants depends upon the balance between respiration and photosynthesis (White 1973). After grazing or clipping, the leaf area left and the age of the leaf tissues largely control a plant's photosyn-

thetic capacity. Leaf blades older than about 28 days generally have a much reduced photosynthetic capacity. A grazing treatment that maintains an abundance of young leaves may therefore give as great or greater carbohydrate storage and herbage production as protection from grazing.

Growth and Development—Along with carbohydrate storage patterns, the developmental morphology of a species may largely determine how and to what degree it may safely be grazed. But aside from work on the phenology of alpine plants (Holway and Ward 1965), little is known about the way meadow species grow.

The apical meristems of vegetative grass shoots retain the capacity to produce new leaves. Such shoots may live from one to several years. When the apical meristems become modified to produce seed, no additional leaf material is produced. Reproductive grass shoots die after seed matures.

Two general groups of grasses—culmless species and culmed species (Hyder 1972)—have been classified on the basis of the developmental morphology of vegetative shoots. If the basal internodes of vegetative shoots show no or little elongation through the season, the species is culmless; if the internodes elongate and thereby elevate the shoot apex, the species is culmed. Other clues help to determine if a species has culmed or culmless vegetative shoots: its growth pattern over the year compared with that of species of known character, and the ratio of reproductive to vegetative shoots. For two culmless grass species, blue grama *(Bouteloua gracillis)* and buffalograss *(Buchloe dactyloides)*, the ratio of reproductive to vegetative shoots was about 1:6 (Rechenthin 1956). For a culmed species, switchgrass *(Panicum virgatum)*, the ratio of reproductive to vegetative shoots was about 2:1.

Culmless species have the apical meristems of vegetative shoots at or near ground level throughout the growing period. The sources of leaf material, therefore, are generally below the level of normal defoliation resulting from grazing. Culmed species, however, elevate the apical meristems of vegetative shoots (as well as of reproductive shoots) at some point in the growing season, thereby exposing them to grazing animals.

These characteristics of grasses influence the appropriate kind of management. Culmless species can often be grazed season long if the degree of use is not excessive. Culmed species may require specialized grazing treatments for highest sustained productivity (Hyder 1972).

Some grasses with culmed vegetative shoots elevate the growing points upon emerging (Rechenthin 1956); in others (Hyder 1972), a number of leaves reach maturity before the apex is elevated. In the latter, the leaves can be grazed before elevation of the apex without stopping leaf growth or development of new leaves. In the early stages, therefore, such species respond similarly to culmless species.

Among the culmless species of grass listed by Heady (1975), only Kentucky bluegrass is common to Sierra Nevada meadows. Bottlebrush squirreltail, which occasionally grows in xeric meadows, is listed as a culmless species also. The growing points of broad-leaved species are in the terminal bud, which is elevated immediately (Rechenthin 1956). Even among forbs, however, differences exist. Common dandelion *(Taraxacum officinale)* has basal leaf primordia that are never elevated (Hyder 1972). This characteristic provides for leaf replacement after removal of the growing points.

Comparatively little is known about the culmless-culmed nature of grasslike plants. We do know that Nebraska sedge is a culmless species (Ratliff 1983). Overwintering mature vegetative shoots of Nebraska sedge have a core of live tissue, and the growing points of vegetative shoots are not elevated during the growing season. Only about one-half of the overwintered shoots become reproductive; including the emerging spring shoots that remain vegetative the first year, the ratio of reproductive to vegetative shoots is about 1:2.

Overwintering shoots of beaked sedge also have a central core of live tissue (Bernard 1973, 1974). All these shoots presumably flower in the following growing season; yet it appears that their growing points are not greatly elevated, if at all, during the previous growing season. Shoots of river sedge *(Carex lacustris)* live for 1 year or less (Bernard and MacDonald 1974). New shoots begin to emerge in July and continue into fall. The overwintered and early shoots of the growing season mature and die. Only late emerging shoots overwinter. Older shoots of river sedge do not have the central core of live vegetation (Bernard 1973). If these two species are compared with Nebraska sedge, it seems likely that beaked sedge is a culmless species and river sedge is a culmed species.

Range Condition

Range condition is defined as "the current productivity of a range relative to what that range is naturally capable of producing" (Range Term glossary Committee 1974, p. 21). Range productivity and forage value are both generally highest when vegetation approaches climax, and lowest when vegetation is far from climax (Sampson 1952).

Forage production and grazing capacity are therefore reflections of range condition—the worse the condition, the lower meadow productivity. Forage depletion was judged to be moderate on 40 percent, material on 42 percent, severe on 15 percent, and extreme on 3 percent of the open forest type in the Western United States (U.S. Dep. Agric., Forest Serv. 1936). Meadows originally produced a large part of the better forage in that type. Depletion on meadows was therefore as great or greater than in the type as a whole.

For the Sierra Nevada, relative reduction in grazing capacity with poorer meadow conditions is approximately constant for all elevations (Crane 1950). Average reductions over seven elevational zones were 35 percent from excellent to good condition, 56 percent from excellent to fair condition, and 75 percent from excellent to poor condition. Given these relationships, a meadow in excellent condition producing 2,500 lb (2,802 kg/ha) of available forage per acre would provide only 1,625 lb (1,821 kg/ha) of available forage per acre in good condition, 1,100 lb per acre (1,233 kg/ha) in fair condition, and 625 lb per acre (700 kg/ha) in poor condition.

Using those differences, reductions from good condition to fair were 32 percent and from good to poor were 62 percent. From fair to poor condition, the reduction was 43 percent. Once damage occurs, reduction of productivity accelerates and is proportionally greater between the lower condition classes. If high livestock production is a goal, it seems essential to manage for excellent meadow conditions.

Fertilization

Fertilization is one means of increasing the productivity of mountain meadows. Improved fertility may increase the more desirable plant species and thereby result in better range condition. Although not usually practiced in National Forests or in National Parks, selected meadows in the Sierra Nevada are fertilized. To my knowledge, only two studies on the effects of fertilization on Sierra Nevada meadows have been reported (Evans and Neal 1982, Leonard and others 1969). Other studies reported are about fertilization of meadows elsewhere, and these are mostly related to hay meadow production. The information, nevertheless, suggests kinds and amounts of fertilizer, methods of application, and is useful in planning fertilization on Sierra Nevada meadows. Major conclusions from meadow fertilization studies have been:

- In general, less productive meadow sites show the greatest proportional quantitative responses.
- Botanical composition may change because of differences in species requirements for various nutrients. Grass and grasslike species generally respond well to nitrogen fertilizer, but legumes increase with application of phosphorus.
- Ammonimum nitrate applied at rates of 270 to 450 lb per acre (303-504 kg/ha) are adequate for most situations and provide about 90 to 150 lb N per acre (100-168 kg N/ha).
- High rates of nitrogen do not appear to result in nitrate buildup to toxic levels, at least on saturated meadow soils.
- Treble-superphosphate applied at rates of 95 to 190 lb per acre (106-213 kg/ha) are adequate for most situations and provide 40 to 80 lb P_2O_5 per acre (45-90 kg P_2O_5/ha).
- Applications of lime may be effective on acid meadow soils; large amounts may, however, be needed to produce a significant response.
- Lower rates of nitrogen application are usually more cost efficient in terms of forage units produced per unit of N applied. But higher rates may be more efficient in terms of units of energy or crude protein produced.
- Cost and return considerations govern whether it is profitable to fertilize a meadow for grazing of livestock.
- Method and timing of fertilizer application may have some effect on subsequent production.

In a meadow along Rock Creek in Sequoia National Park, herbage production on an unfertilized site was 1,205 lb per acre (1,350 kg/ha) (Giffen and others 1970, Leonard and others 1969). With 250 lb per acre (280 kg/ha) of ammonimum nitrate, the site produced 2,185 lb per acre (2,450 kg/ha). And with a like amount of gypsum, the site produced 1,338 lb per acre (1,500 kg/ha). In addition, 100 lb per acre (112 kg/ha) of copper sulfate applied as a foliar spray produced greener vegetation on peat soils, suggesting that application of minor elements may increase production in some situations.

Phosphorus alone or in combination with nitrogen or nitrogen and sulfur increased yields of Blando brome *(Bromus mollis)* grown in soil from two meadows on the Sierra National Forest (Evans and Neal 1982). Sources of nutrients were ammonium nitrate, phosphoric acid, and sodium sulfate.

At rates of 50 lb per acre and 200 lb per acre (56 kg/ha and 224 kg/ha) nitrogen did not significantly increase Blando brome yields. Phosphorus at 50 lb per acre did not increase yield over the control, but at 200 lb per acre the yield was increased 6 to 12 times. Combined, the low rates of nitrogen and phosphorus produced yields equal to those from the high rate of phosphorus alone. The same result occurred with the low rate of nitrogen and the high rate of phosphorus and with the high rate of nitrogen and the low rate of phosphorus. Maximum yields were obtained with high rates of nitrogen and phosphorus on one of the meadows soils. Maximum yields from the other meadow soil were obtained when sulfur at 100 lb per acre (112 kg/ha) was included with the high rates of nitrogen and phosphorus. Maximum yields from both meadow soils were 24 times greater than those from the controls.

In British Columbia, yield on a Kentucky bluegrass site was increased an average of 1,100 lb per acre (1,233 kg/ha) by applying 100 lb N per acre (112 kg N/ha) (Mason and Miltmore 1969). Ammonium nitrate fertilizer was used.

On another British Columbia meadow, seven of nine fertilizer treatments produced greater yields than the controls (McLean and others 1963). Botanical composition of that meadow was largely sedges with beaked sedge as a primary component. Although the seven treatments did not differ significantly, the data provided some valuable information. A 10-20-10 fertilizer formulation applied at a rate of 400 lb per acre (448 kg/ha) produced 920 lb per acre (1,031 kg/ha) more forage than the control, which produced 860 lb per acre (964 kg/ha). Treatments containing phosphorus produced an average of 700 lb per acre (785 kg/ha) more forage. Hydrated lime alone produced 640 lb per acre (717 kg/ha) more forage than the control. The total amount of lime applied was 3,000 lb per acre (3,362 kg/ha). The response to lime was thought to result from its neutralizing effect on acid soil. The 16-20-0, 0-19-0, and 11-48-0 formulations produced 627, 600, and 200 lb per acre (703, 672, and 224 kg/ha) more forage than the control. The two treatments containing potassium produced the highest average yields (1,700 lb/acre or 1,905 kg/ha). The conclusion was that application of lime, phosphate, or complete fertilizers increases yields.

In the Nebraska Sandhills, ammonium nitrate was applied at rates of 0, 40, and 80 lb N per acre (0, 45, and 90 kg/ha); and treble superphosphate at rates of 0, 40, 80, and 160 lb P_2O_5 per acre (0, 45, 90, and 180 kg/ha) (Russell and others 1965). Alone, nitrogen had little effect on production but interacted

with phosphorus. At the highest rates of phosphorus, the average yields with 40 lb and 80 lb N applications ranged from 18 to 50 percent higher than without nitrogen. The greatest relative response was on the lowest yielding site. Actual yield increases ranged from about 500 lb per acre (560 kg/ha) to 800 lb per acre (897 kg/ha), with the lowest yield from the poorest site. Phosphorus alone increased yields. Average increases in yields over the controls were about 18, 28, and 24 percent (320, 505, and 440 lb/acre or 359, 566, and 493 kg/ha) for the 40-, 80-, and 160-lb rates.

Ammonium nitrate was applied at rates of 0, 20, 40, and 60 lb N per acre (0, 22, 45 and 67 kg/ha) and treble superphosphate was applied at rates of 0, 40, 80, and 120 lb P_2O_5 per acre (0, 45, 90, and 135 kg/ha), on meadows in eastern Oregon (Cooper and Sawyer 1955). No significant interactions between nitrogen and phosphorus levels were found. The 60 lb per acre rate of N increased yields by a ton (2,000 lb or 907 kg) over the control, which produced 3,500 lb per acre (3,923 kg/ha). Nitrogen at that rate had no carry-over effect. Phosphorus increased yields 660 lb per acre (740 kg/ha). Rates of P_2O_5 greater than 40 lb per acre had no significant advantage. A carryover effect was observed with phosphorus. The second year after fertilization, yields were 400 lb per acre (448 kg/ha) higher than those of the control. In the same area, applications of 50, 100, 150, and 200 lb N per acre (56, 112, 168, and 224 kg N/ha) produced yields 43, 67, 82, and 100 percent greater than the controls (Nelson and Castle 1958).

Ammonium sulfate was applied on a native flood meadow in Oregon (Gomm 1980). Application rates were 0, 98, 196, and 393 lb N per acre (0, 110, 220, and 440 kg N/ha). All three levels of nitrogen resulted in greater yields than the control. The yields were 3,745, 6,218, 7,179, and 8,571 lb per acre (4,198, 6,969, 8,046, and 9,607 kg/ha).

A meadow at 10,200-ft (3,109-m) elevation in Utah was fertilized for 3 consecutive years with ammonium sulfate and treble superphosphate (Browns 1972). Treatments in pounds per acre of N and P were 30 N, 60 N, 30 P, 60 P, 30 N + 30 P, and 60 N + 60 P. The greatest production was given by the combined high-rate treatment, followed by the combined low-rate treatment and the high-rate nitrogen treatment. Increases in production over the control (965 lb/acre or 1,082 kg/ha) for those treatments were 628, 525, and 519 lb per acre. Significant carryover effects were not found the third year after fertilization. Visual differences were, however, still apparent between the control and treatment plots.

Ammonium nitrate at rates of 0, 80, and 160 lb N per acre (0, 90, and 179 kg N/ha) was applied annually to meadows in Wyoming (Lewis and Lang 1957). Average yields for all eight grasses studied were 1,600, 5,800, and 7,400 lb per acre (1,793, 6,501, and 8,294 kg/ha) for the 0-, 80-, and 160-lb rates. Some effect of the 160-lb rate was observed in regrowth after harvest, but no such effect was observed for the 80-lb rate.

The percent of calcium and phosphorus in the grasses was increased by nitrogen fertilizer. Fertilized plants had more seedstalks than did the controls and were darker green. Plants fertilized at the 160-lb (179-kg) rate, however, had fewer seedstalks than those fertilized at the 80-lb (90-kg) rate and were later in maturing. And at the higher rate, plants were more easily lodged or beat down (Lewis and Lang 1957).

Large variations in yield exist between meadows, and manuring has an equalizing effect on average yields (Klapp 1962). In assessing responses to manuring, effects on species composition need to be assessed. Hay yields of more than 8,900 lb per acre (9,975 kg/ha) are possible with manuring on farm meadows in Europe. On wild meadows, the relative effect of manuring is even greater. Marked changes in species composition are, however, associated with the effect.

Changes in botanical and chemical composition of forage plants and yield responses should be measured for evaluating the effects of fertilization (Russell and others 1965). Alone, nitrogen fertilizer stimulated grasses and grasslikes, causing a reduction in the amount of clover. Phosphorus when used alone stimulated the legumes. Combined, nitrogen and phosphorus stimulated the grasses, grasslikes, and legumes to produce the greatest yields. Increases in clover with phosphorus applications also were reported (Cooper and Sawyer 1955).

Nitrogen applied in fall was profitable when the forage was harvested for hay (Workman and Quigley 1974). When the forage was harvested directly by livestock, fertilization was not profitable. Small differences in prices could, however reverse the latter conclusion. The 50-lb N per acre rate (Nelson and Castle 1958) was the most efficient in increased hay production—32 lb (14.5 kg)—per pound of fertilizer. Although the return in forage per dollar of N was greater at the 80-lb N per acre rate (Lewis and Lang 1957), the return in crude protein was greater at the 160-lb N per acre rate of application.

Compared to surface application, drilling treble superphosphate to depths of 3 or 4 inches (8-10 cm) reduced yields on subirrigated meadows in Nebraska (Moore and others 1968). The differences were largely attributed to the killing of plants from drilling rather than to fertilizer placement. In the year of treatment, yields were 2,500 lb per acre (2,802 kg/ha) by drilling and 2,950 lb per acre (3,306 kg/ha) by surface application. The differences were less the next year, and one site produced greater yields on the drilled treatment. That result was thought to result from fixation of phosphorus applied to the surface.

Nitrate concentrations as low as 0.21 percent can kill cattle (Gomm 1979). When meadow plants are grown in saturated soil, the tissues apparently do not accumulate toxic nitrate concentrations. When grown in unsaturated soil, however, nitrate can accumulate to toxic levels. At normal application rates, toxic level accumulations are unlikely.

Although fertilization of meadows appears to be an attractive way to increase productivity, a few cautions are advisable:

- Decide the purpose of the fertilization and state it clearly. If the purpose is to increase red meat production, plan for the efficient, full use of the increased forage. If the purpose is to add organic matter to the ecosystem for range improvement, plan to protect the meadow from grazing.
- Test the soil to determine which fertilizer components and how much is most effective. Do not waste money and

effort by applying the wrong fertilizer mix or by using more than is needed.

- Consider the micronutrients. Low meadow productivity may be associated with deficient or toxic levels, or both, of the micronutrients.
- Fence and manage a fertilized meadow apart from the rest of a grazing allotment to prevent overuse by livestock and wildlife.
- Continue fertilization to maintain the fertility level. Permanent increases in permitted use should not be based on the increased forage, unless a continuing fertilization program is assured.

Utilization

Mountain meadows of the Sierra Nevada historically have been grazed by domestic and wild herbivores. Continued overuse has resulted in and can be expected to result in meadow deterioration. The key to continued productivity of mountain meadows is proper utilization—"a degree and time of use of current year's growth which, if continued, will either maintain or improve the range condition consistent with conservation of other natural resources" (Range Term Glossary Committee 1974, p. 21).

A most reliable indicator of range condition is the amount of mulch or residue on the ground (Sampson 1952). Residue must be left after grazing for the range to sustain production (Stoddart and others 1975). For grassland vegetation, several authors have reported benefits from leaving some amount of the current herbage (Bement 1969, Bentley and Talbot 1951, Heady 1956, Hooper and Heady 1970, Hormay 1944). The 50-50 rule (graze half—leave half) is a good guide to conservative range use (Frandsen 1961). Implicit in such findings and statements is the idea that some herbage must be left to decompose. But the key question is this: What proportion of the current herbage production should be left to decompose (left unconsumed by the herbivore component) to maintain or improve meadow ecosystems?

I have suggested that the amount left should equal the average proportion that decomposes annually (Ratliff 1976, 1980). Yet I have no evidence to support that suggestion. If it is valid, then to maintain meadow productivity, utilization cannot (on the average) exceed $[(A - AK)/A]\ 100$, where A represents annual production or inputs to the mulch layer, and K represents the proportion decomposing in one year (Jenny and others 1949). The concept is this: when annual inputs exceed decomposition, the mulch layer increases and soil organic matter content stabilizes. When annual inputs are less than decomposition, the mulch layer and soil organic matter is consumed by the decomposers, resulting in instability.

For five sites in Sequoia National Park (Hair Sedge, Lake Shore, Chagoopa, Big Arroyo, and Sky Parlor), the K values were estimated to be 64, 51, 57, 55, and 66 percent (Ratliff 1980). Respective degrees of proper utilization at those sites would be 36, 49, 43, 45, and 34 percent. The lower values are for the driest and wettest of the five sites. It seems possible that sites having extreme environmental influences may not be able to sustain as heavy utilization as sites having more moderate environments. And the 50-50 rule cannot be considered a safe utilization guide for all meadow sites of the Sierra Nevada.

Given that a site is not deteriorated I suggest that utilization of meadow sites in the raised convex, hanging, lotic, and xeric hydrologic series should not exceed 35 percent of the average annual herbage production. An average utilization of 45 percent for meadow sites of the normal and sunken-concave hydrologic series should maintain productivity. These suggestions compare with actual utilizations at Markwood (44 percent) and Exchequer (37 percent) meadows on the Sierra National Forest (Clayton 1974). Average utilization of about 59 percent was reported for meadows at Blodgett Forest (Kosco 1980). Because no change in productivity was observed over the study period on Blodgett Forest, either the current level of utilization was acceptable or production had stabilized at a level lower than in pristine times (Kosco 1980).

MANAGEMENT PROBLEMS

Four major problems can plague a meadow manager: animal activities, lodgepole pine, fire, and gully erosion. In creating problems, use of meadow resources for livestock production stands out. Livestock grazing creates problems when defoliation, preferential grazing, trampling, and mineral redistribution are not in harmony with plant needs for growth and reproduction. Overgrazing may accentuate the ill effects of rodent use, create conditions favorable to lodgepole pine invasion, alter meadow-fire relationships, and accelerate erosion. Well-managed livestock grazing of meadows should produce no lasting negative effects.

Many acres of meadow appear to have been lost to forest since about 1900—after the period of heavy use by sheep. The loss has put added pressure on the remaining meadow area to produce what humans want from meadows. Fire in dry, ungrazed meadows may be a significant influence in maintaining stable forest-meadow boundaries. Fires on the watershed, however, are likely of more frequent influence to meadow ecology through increased water flow and subsequent sedimentation. Lowering the water table induces change in meadow vegetation. The breaking up or depletion of the protective sod cover and activities on the watershed may alter the threshold levels of erosive forces, resulting in erosion and gully formation. Gully erosion must be controlled and the water table raised to maintain or restore a meadow.

Animal Activities

The effect of a lone animal—or person—crossing a meadow is insignificant. But multiply the crossing tenfold, a

hundredfold, or a thousandfold, and the net effect on the meadow could be highly significant.

When an external influence (a stress) results in a measurable and lasting change (a strain) to a meadow community or its soil, or both, the effect is significant (Sharma and others 1976). A strain may be elastic (reversible) or plastic (irreversible). Effects associated with elastic strains last for relatively short periods and are usually nondestructive. Effects associated with plastic strains may last for many years and usually are destructive. Recovery to the original plant community may be unlikely because of altered conditions.

An occasional season of overuse usually produces an elastic strain. An effect that is obvious at the close of the grazing season should be undetectable the next season. Continued overuse resulting in an overgrazed condition produces a plastic strain. Climax species may be replaced by less desirable plants, eroded soil, or both.

Wildlife and humans can be agents of stress. But livestock are the agents of stress that come to mind when effects on meadows are discussed. Nine "grazing factors" (intensity, season, and frequency of defoliation; selectivity in grazing; plant, mineral, and animal redistribution; mineral cycling; physical impacts; and animal behavior) interact to affect decisions about range use (Heady 1975). Effects of defoliation, preferential grazing, trampling, mineral redistribution, and burrowing by rodents on meadows are discussed here.

Defoliation

Defoliation of meadow species by grazing animals is regarded as a major cause of meadow deterioration and of rangeland in general. The degree, frequency, and timing of grazing or other defoliation affects a plant's ability to produce herbage, reproduce, and survive. Those factors, therefore, are fundamental to determining proper use for herbivore production and setting utilization standards. "Any factor affecting photosynthesis or utilization of carbohydrates for respiration or growth will affect the level of plant reserves" (Cook 1966). Effects of defoliation are associated also with removal of meristematic tissues (Heady 1975), and for most species, defoliation reduces root growth (Younger 1972).

Although a large body of literature is available on the effects of defoliation on plants, information about such effects on meadow plants is scant. To my knowledge, the only published work on defoliation effects for meadow species of the Sierra Nevada is that of DeBenedetti (1980). He found that clipping the herbage to a 1-inch (2.5-cm) stubble height reduced total nonstructural carbohydrates in roots of shorthair sedge, shorthair, and *Carex scopulorum* by 20 to 40 percent.

Although restricted to a single intensity of clipping, DeBenedetti's study involved four clipping regimes continued over a 3-year period. Herbage yield data from the study is still being analyzed.

The response of western false-hellebore *(Veratrum californicum)* to cutting at ground level during early spring emergence was studied (McDougald 1976). Three years of such treatment virtually eliminated the species from the study plot. Two years later the plants had recovered only slightly. We know that defoliation is generally most damaging to a plant when carbohydrate reserves are at their low point, and that carbohydrate reserves are generally lowest after growth begins. The response of western false-hellebore to early spring cutting may be associated with a low carbohydrate reserve at that time and perhaps early elevation and removal of the growing point.

Reductions in hay yields from native flood meadows in eastern Oregon were blamed on loss of soil fertility rather than on time or height of cutting (Cooper 1956). After 4 years of cutting, no significant differences owing to treatment were found in either yields or species composition.

Plots on three meadows of the Bighorn National Forest in Wyoming were clipped to either a 1-inch (2.5-cm) or 3-inch (7.6-cm) stubble height every 2 weeks (Pond 1961). The controls were clipped to a 1-inch (2.5-cm) stubble at seasons end, to estimate total production. Kentucky bluegrass density increased with all treatments, except the control on one meadow which showed a slight decrease. Tufted hairgrass decreased in all meadows with clipping to 1 inch (2.5 cm) every 2 weeks, but remained the same or increased with the other treatments. These findings support the placement of tufted hairgrass as a decreaser and Kentucky bluegrass as an increaser. On the two meadows where it grew, redtop *(Agrostis alba)* either disappeared or was greatly reduced by the most severe treatment. It decreased slightly or remained the same under the other treatments. Alpine timothy *(Phleum alpinum)* was reported for one meadow, where it remained the same on the control and with the 3-inch (7.6-cm) cutting treatment. Beaked sedge disappeared with the 1-inch (2.5-cm) cutting treatment on two of the meadows. It was reduced by the 3-inch (7.6-cm) treatment but remained the same on the control plots. On the other meadow, beaked sedge disappeared even with the control treatment, but little of it was growing to start with. Ovalhead sedge *(Carex festivella)* decreased with both cutting treatments, but showed a slight increase on the control plot in one meadow. Baltic rush *(Juncus balticus)* density declined with all treatments, including the control. Density of forbs declined with all three treatments on two of the meadows, but increased with all treatments on the third meadow. All or most species mentioned grow on meadows of the Sierra Nevada, and it is obvious that a cutting or grazing intensity favorable to one species may not be favorable to another.

Beaked sedge was among the species in a study of six frequencies of clipping to a stubble height of 2 inches (5.1 cm) in British Columbia (McLean and others 1963). The clipping schedules were every 2, 4, and 8 weeks; every 6 weeks with a 2-week delay into the grazing season; every 4 weeks with a 4-week delay; and at the end of the grazing season. Individual species' responses were not reported, but no visual changes in composition were seen. Frequent clipping (every 2 to 4 weeks from the start of grazing) lowered subsequent forage yields. In some instances, plots not clipped late in the season owing to insufficient regrowth gave their highest yields the next year. This suggests that resting the meadow near the end of the

season allows the sedge species to build their carbohydrate reserves and is in agreement with what we now know about Nebraska sedge (Steele 1981, Steele and others 1984).

Preferential Grazing

Grazing preference is "selection of certain plants over others" and selective grazing is "grazing of certain plant species on the range to the exclusion of others" (Range Term Glossary Committee 1974, p. 12, 25). I use the term "preferential grazing" (Ratliff 1962) to describe the grazing of certain range areas and certain plants or plant species, or both, in similar yearly patterns. Because of changing palatabilities as the season advances, most plant species are grazed more heavily at one time than another. And preferential grazing is a major cause of range deterioration (Hormay and Talbot 1961). This practice, when combined with too severe or too frequent defoliation, eventually reduces abundance of the preferred species. The only sure way to counter the effects of preferential grazing is to restrict grazing by livestock under some form of rotation. The rotation scheme must be keyed to the growth and reproduction requirements of the preferred species. The land manager therefore needs information on meadow site and plant preferences of grazing animals.

Grazing preferences by animals have been reported by Bell (1973), Heady (1975), Sampson (1952), and Stoddart and others (1975). Horses are the most selective among domestic animals, eating more rough forage than other classes of livestock. Cattle prefer grasses, browse, and forbs, in that order. Sheep use grass in quantity but make fuller use of forbs than cattle. Goats and sheep are best adapted to browse ranges. Elk prefer sedges and grasses but will graze forbs in summer. Grass and grasslike plants are necessary in the diet of bighorn sheep as well, although they use forbs in spring and summer. Forbs and browses constitute the primary food year around in the diet of deer.

Fertility, topography, and wetness help determine preferences. Fertilized and burned areas are preferred by all kinds of grazing animals; consequently, small fertilized areas may be overused and damaged. Cattle and horses tend to prefer level or rolling range. Sheep and goats are well adapted to steep topography. Such preferences may be related to where the animals were raised. Animals raised in rough country frequently make better use of steep areas than those raised in flat country. Sheep tend to avoid the wetter portions of meadows. Cattle graze farther out on a meadow as it dries and wade to graze selected species.

Preferential grazing in the Sierra Nevada has been studied, but only a few studies concern meadows. Deer selected sedges and rushes of "seepages" on winter range (Evans 1976). They browsed most heavily from late summer through early winter and during the spring growth period (Bertram 1982). Grasses were the principal food in fall on winter ranges and during spring migration. Forbs were eaten primarily from midwinter through early fall.

Counts of bites were used to study deer food habits at Markwood Meadow on the Sierra National Forest (Chesemore and others 1976). Tame yearling deer were observed from June 16 to August 18, 1975. The yearlings ate 80 food items. Forbs dominated the diet. Of the total diet, 75 percent was forbs, 12 percent grasses and grasslikes, 10 percent shrubs, and 3 percent other species. Among the forbs, sheep sorrel *(Rumex angiocarpus)* was predominant and was most important in the total diet as well. Sheep sorrel made up 25 percent of the diet at the start of the study, reached a peak of 29 percent in early July, and declined to 13 percent in mid-August. Other forbs contributing significantly to the diet were sticky cinquefoil *(Potentilla glandulosa)* (14 pct), American bistort (11 pct), and Yosemite ivesia *(Ivesia unguiculata)* (9 pct). American bistort and Yosemite ivesia were preferred by cattle and deer (Chesemore and others 1976). American bistort and sticky cinquefoil are considered increaser species. Sheep sorrel and Yosemite ivesia are considered invaders of meadows. Pointed rush *(Juncus oxymeris),* the most significant of the grasses and grasslike species, made up 6 percent of the total diet. Sedges comprised 4 percent and grasses 0.6 percent of the diet. The sedges, rushes, and grasses were grazed by cattle and deer.

Initially, cattle preferred the drier portions of Markwood and Tule meadows on the Sierra National Forest (Pattee 1973). As the meadows dried, cattle used the other portions. Tule meadow has a wet center and relatively dry edges. Markwood meadow has a dry center and relatively wet edges. Cattle spent 91 percent of their time in the center of Markwood meadow and 100 percent of their time on the edges of Tule meadow.

Species preferences by cattle and deer grazing on meadows could not be determined by direct observation because of vegetation density (Pattee 1973). Although dealing mainly with forage utilization, Clayton (1974) provided data on preferences. His study included three meadows—Markwood, Exchequer, and Three Springs—on the Sierra National Forest. In each meadow, each of 10 plots were sampled two or three times with 150 points. Total numbers and grazed numbers of plants or shoots of selected species were given. Using chi-square, I tested the hypothesis that the ratio of grazed to ungrazed plants or shoots was independent of species. This is equivalent to saying that the species are grazed in proportion to their abundance in the sample. Each set of data was tested separately. For each meadow and sampling for which independence was rejected *(table 11),* I computed chi-square for the individual species. Only species with one or more significant chi-squares are presented *(table 12).* The species names are as given by Clayton (1974). Although, the sedges he names can grow in meadows of the Sierra Nevada, what he has called sand sedge *(Carex aquatilis)* is Nebraska sedge according to Chesemore and others (1976). And I suggest that C. *vernacula* is abruptbeak sedge *(C. abrupta).*

In general, use of the sedges, tufted hairgrass, and tinkers penny was either less than or the same as expected by chance *(table 12).* Use of California oatgrass, *Deschampsia elongata,* Jeffrey shooting-star, *Rumex acetosella,* and cow clover *(Trifolium wormskioldii)* was either more than or the same as expected by chance. The numbers of plants or shoots observed were small; nevertheless, those species were evi-

Table 11—*Chi-square values for tests of independence between the ratio of grazed to ungrazed plants and plant species at three meadows in the Sierra Nevada, California, by sampling dates*

Sampling dates	Meadow		
	Markwood	Exchequer	Three Springs
	Chi-square values[1]		
1971:			
July	2.17 (3)		
August			24.25 (8)
1972			
June	44.86 (7)		
July		38.55 (9)	73.57 (14)
August	53.17 (12)	31.05 (11)	11.68 (8)

Source: Clayton 1974

[1]Degrees of freedom (df) are in parentheses.

dently selected by grazing animals. American bistort, elephant heads *(Pedicularis groenlandica),* and pointed rush belong to this group also. But they were relatively abundant. Bolander's sunflower *(Phalacroseris bolanderi)* shows the effects of palatability changes. It was not selected at the July 1972 sampling but was selected by the August sampling.

Cattle had not grazed Markwood Meadow at the June 1972 sampling. The results from that sampling, therefore, represent early use by wildlife. Again, the sedges were not selected. The other species were grazed more than or the same as expected.

Preference by horses and mules is related to degree of hunger (Strand 1979a). As the length of the grazing period increased they became more selective. Mules were observed to show greater degree of preference than horses. One mule repeatedly moved some distance to obtain *Elymus glaucus* (blue wildrye). During trips into the Minarets Wilderness Area in late August of 1978 and 1981, I observed that horses select and closely graze Sierra bilberry.

Trampling

Trampling refers to "the damage to plants or soil brought about by movements or congestion of animals" (Range Term Glossary Committee 1974, p. 28). As used here, trampling also refers to damage to plants or soil by movements or congestion of people. Its effects are of great concern to land managers because of damage to the resource. Trampling damage to meadows results from two main effects: compaction that alters soil structure, and cutting of the sod that leaves bare spots or mud holes. Both effects can result in soil loss and changes in species composition.

Although certain meadow areas have not been noticeably altered after many years of heavy use, others cannot tolerate even light use without being adversely affected (Strand 1979a). The major meadow characteristics that determine the degree or extent of trampling damage are elevation, slope, and hydrology. Generally, as elevation increases, meadow recovery rate decreases; as slope increases, meadow fragility increases; and, the hydrologic regime largely determines the species composition and soil properties. The species present influence the kinds of animals using the meadow and when they use it.

The primary effect of compaction is to alter the surface structure of the soil, thereby encouraging conditions favorable for erosion. Compaction increases bulk density, reduces total and noncapillary pore space, and lowers infiltration and percolation rates (Lull 1959).

Compaction is governed by the degree that stress overcomes soil resistance to deformation (Lull 1959). As resistance is overcome, soil particles and aggregates become packed—reducing pore space, increasing bulk density, and increasing resistance. When resistance and stress are in equilibrium, compaction ceases. Counterstresses of swelling and shrinking, with changes in soil moisture and temperature, largely determine how long a soil remains compacted.

Soil resistance to deformation is governed by the relationships among moisture content, texture, structure, density, and organic content (Lull 1959). Generally, resistance to stress and compaction lessens as soil moisture content increases. But maximum bulk densities are produced by compaction at a moisture content midway between the wilting point and field capacity. Maximum bulk densities are lower in clays than in gravely soils, but soils with widely different size grains compact more than soils with more uniform size grains. Soils with strong structures have durable peds, natural soil aggregates. They have lower bulk densities, are more permeable, and are more resistant to compaction than like-textured soils with weak structures. Although freezing and thawing tend to loosen compacted soils, they tend to compact soils by destroying soil structure. Resistance becomes stronger as compaction increases the bulk density. Organic matter increases resistance by raising the moisture content needed for compaction. Also, organic matter binds soil particles into aggregates and cushions the mineral soil surface.

Livestock and humans can exert sufficient pressure to compact meadow soils. Static ground pressures exerted by livestock were estimated (Lull 1959):

	Weight		Static ground pressures	
	lb	*kg*	*lbs/inch*^{2}	*kg/cm*^{2}
Livestock:				
Sheep	120	54	9.2	0.6
Cattle	1350	612	23.9	1.7
Horses	—	—	20.0 to 57.0	1.4 to 4.0

A 150-lb (68-kg) person whose shoes have a bearing surface of 24 inch^2 (155 cm^2), exerts a static ground pressure of 6.2 lb per inch^2 (0.4 kg/cm^2). During normal movement, the pressures exerted may be doubled or quadrupled because all the weight may be put on one hoof or foot. Presumably, when running, the pressures exerted are greater. Unless the soil is dry, or otherwise firm enough to support these pressures, the soil structure deforms and compaction results.

A compacted soil restricts root penetration. On a highly compacted meadow, the fibrous roots of perennial grasses and rhizomes of grasslike plants may not be able to develop normally. Plants with strong taproots or shallow rooted annuals may therefore become dominant.

Table 12—*Preferences of grazing animals for individual plant species on three meadows in the Sierra Nevada, California, at four sampling dates*[1]

Species and meadow	August 1971				June 1972				July 1972				August 1972			
	N	O	E	χ^2	N	O	E	χ^2	N	O	E	χ^2	N	O	E	χ^2
Carex aquatilis:																
Markwood					56	1	9.9	9.7					170	68	102.0	28.4
Exchequer									56	7	11.6	NS	61	12	12.7	NS
Three Springs	44	17	20.6	NS					67	15	25.1	6.5				
Carex teneraeformis:																
Exchequer									3	3	0.6	11.4				
Carex vernacula:																
Markwood					44	2	7.8	5.2					15	7	9.0	NS
Exchequer													78	9	16.3	4.1
Three Springs	16	8	7.5	NS					24	4	9.0	4.4				
Danthonia californica:																
Markwood													55	38	33.0	NS
Three Springs									4	4	1.5	4.2				
Deschampsia caespitosa:																
Markwood					22	7	3.9	NS								
Exchequer									7	3	1.5	NS	48	11	10.0	NS
Three Springs	56	18	26.3	4.9					42	19	15.7	NS				
Deschampsia elongata:																
Markwood					6	2	1.1	NS								
Three Springs									3	3	1.1	5.0				
Dodecatheon jeffreyi:																
Exchequer													2	2	0.4	7.6
Three Springs									6	4	2.2	NS				
Hypericum anagalloides:																
Exchequer													5	1	1.0	NS
Three Springs									17	1	6.4	7.2				
Juncus oxymeris:																
Markwood					9	4	1.6	4.4					339	216	203.5	NS
Exchequer									48	20	10.0	12.7	140	39	29.2	4.2
Three Springs	34	25	16.0	9.7					39	32	14.6	33.1				
Pedicularis groenlandica:																
Markwood													24	22	14.4	10.0
Three Springs									2	1	0.8	NS				
Phalacroseris bolanderi:																
Markwood													11	5	6.6	NS
Exchequer									49	2	10.2	8.3	22	10	4.6	8.1
Polygonum bistortoides:																
Markwood					30	11	5.3	7.5					34	25	20.4	NS
Exchequer									3	1	0.6	NS				
Three Springs	5	1	2.4	NS												
Rumex Acetosella:																
Markwood					1	1	0.2	4.7								
Trifolium wormskioldii:																
Markwood					2	2	0.4	9.3								

Source: Clayton (1974).

[1]N = Number of shoots or plants encountered in sample.
O = Observed number of shoots or plants which had been grazed.
E = Expected number of shoots or plants grazed
NS = Not significant—chi-square < 3.84 ($\chi^2_{0.05}$ with 1 degree of freedom).

Compaction may not be apparent in meadows having a high content of organic matter in the surface layer. In such meadows, owing to the resistance of organic matter, the surface layer may show little or no compaction. But soil below the surface may be compacted and restrict water movement and roots. Such compaction must in time become evident through changes in composition of the vegetation and breakdown of the sod.

Trampling can alter the chemistry of the soil. In Sequoia National Park, the soil solution at the surface was more acid in and adjacent to the trail tread through a wet meadow than in the undisturbed portions (Leonard and others 1968). The pHs at the junction of the John Muir and Crabtree Meadow trails showed effects of trampling:

Location or material:	pH
Slurry at bridge (trampled)	5.2
Muddy water in trail (trampled)	5.4
Below trail (undisturbed)	6.2
Forb meadow between trails	5.4
Water in trail tread	5.4
Mud in trail bottom	5.4
Carex seep (undisturbed)	5.8
Carex seep (trampled)	5.4
Incoming water to area	6.2

Also, the pH could be dropped as much as one unit by trampling the meadow surface. The pHs in Rock Creek and Crabtree meadows were different before and after trampling (Leonard and others 1968):

	pH	
Meadow and site:	Before trampling	After trampling
Rock Creek		
Near rock	6.1	4.3
Rusty sedge area	5.2	4.4
Crabtree		
Seepage at Cabin Spring	6.8	6.5
Eroded muddy area in seepage	6.4	5.4
Sphagnum moss patch	7.0	5.8

One effect of pH change associated with trampling may be to favor species that tolerate more acid conditions over those species that are less tolerant. Trampling, however, affects vegetation in other ways as well. The most obvious signs of trampling are holes punched in meadows by livestock. This effect of trampling cuts the sod and, if concentrated, kills the vegetation, resulting in a trail or a large mud hole.

Generally, as the number of passes over an area increase, compaction will increase up to a point of maximum density (Lull 1959). This, along with damage to the sod, is why trails develop across meadows.

For the most part, low levels of people trampling do not permanently affect meadow vegetation (Palmer 1979). Height growth of a shorthair site, however, was reduced the year after 600 tramplings. And a *Phyllodoce breweri* (red mountain heather) community showed no recovery 2 years after being trampled 210 times. Low trampling levels (only five passes) will crush and flatten most plants, making somewhat of a path. This can lead others to follow the same route and, therefore, eventually damage the vegetation and produce a trail. People, like cattle, tend "to follow the leader."

The degree and persistence of trampling damage by packstock are related to soil wetness at the time of trampling and the amount of trampling (Strand 1979b). On a short-hair sedge site (xeric hydrology), 100 passes by horse and rider made a ring that was visible 1 but not 2 years later. On a wet meadow site (with standing water), only 25 passes were enough to trample 90 percent of the vegetation into the mud. Foliar cover was reduced 75 percent 1 year later and 30 percent 2 years later.

Cattle create trails between meadows. They make lengthwise trails in long meadows. Seldom is a cow trail built across a meadow: on reaching a meadow, cattle normally disperse for grazing; on leaving a meadow, they gather to the trail from dispersed points. I suggest, therefore, that trails across meadows are made by people. Many such trails likely originated when people on foot or horseback followed a game, sheep, or cow trail to a meadow edge and then proceeded across to pick up a trail on the other side.

This is not to deny that free-to-roam livestock cause trampling damage. They can and do. Such damage may be especially obvious on hillside bogs where hoof cuts may not heal for several years. Sphagnum rich meadows like Charlotte Lake Meadow in Kings Canyon National Park are especially subject to this kind of damage (Strand 1979a).

To reduce compaction and trails in meadows, I suggest:

- Adjusting use, particularly of high-elevation meadows and soft meadow edges, to periods when the soil is firm enough to support grazing livestock.
- Closing (fencing) sensitive sites to livestock grazing and other people uses.
- Fertilizing meadows or sites most resistant to trampling damage to attract livestock and wildlife from sensitive sites.
- Locating salt grounds well away from meadows to improve livestock distribution.
- Routing trails to keep transient livestock and people off meadows.
- Instructing people to walk abreast rather than in line if they must cross a meadow.
- Instructing people to use a different route from camp each time they cross a meadow to get water. Moderate grazing to reduce the height of the vegetation may be needed to achieve this in some situations.

Trampling of meadows, at least in moderate degree and in some situations, may not be all bad. Animals and people may transport rhizomes and seed in mud adhering to their hooves or boots and effectively replant a disturbed or degraded meadow. Some compaction may reduce frost damage to meadow vegetation.

Trampling to some degree will occur so long as livestock and people are allowed on meadows. For practical purposes, therefore, trampling cannot be prevented. But the land manager needs to take measures to reduce its damaging effects.

Mineral Redistribution

Redistribution of minerals is a natural consequence of grazing. It is the least likely of the nine grazing factors (Heady 1977) to be altered by management practices. Nevertheless, it is frequently considered to have adverse effects on meadows.

Relatively small amounts of mineral nutrients consumed by herbivores are actually lost to the ecosystem (Pieper 1977). Of the minerals consumed, these amounts are estimated to be returned to the ecosystem in feces and urine: nitrogen (89.7 pct), phosphorus (61.9 pct), potassium (96 pct), calcium (86.5 pct), and magnesium (89 pct). Mineral losses resulting from livestock grazing a meadow, pasture, or allotment are therefore of little significance.

Redistribution of minerals may be of greater significance. "The net effect of herbivores on nutrient cycling is to remove nutrients from some areas and to concentrate them in others" (Pieper 1977, p. 266). Mineral nutrients are redistributed when animals consistently feed in one place and deposit excreta some distance away at "focal points . . . water, salt, feeding areas, bed-grounds, and shade" (Heady 1975, p. 91). Percentages in dung of the total amounts in cattle and rodent dung and urine were estimated to be nitrogen (12 pct), phosphorus (77 pct), potassium (6 pct), calcium (72 pct), and magnesium (85 pct) (Pieper 1977). Each cow dung pat covers from 0.4 ft^2 to 0.8 ft^2 (372 cm^2 to 743 cm^2), but the total area affected is about 2.6 ft^2 (0.25 m^2) (Castle and MacDaid 1972).

Cattle defecate 10 to 16 times per day and urinate 8 to 12 times per day (MacLusky 1960). Together, the dung and urine voided per day affects about 8 m^2 (Heady 1975). Unless livestock defecate and urinate on meadows in direct proportion to the amount of herbage consumed there, some redistribution of minerals will occur.

Conditions conducive to rapid decomposition—warm, moist environments—are usually present in meadows during the grazing season. Dung on seeded and fertilized pastures crumbled in 59 days and disappeared in 114 days (Castle and MacDaid 1972). Dung deposited in July disappeared more quickly than that deposited in June. Fertilization did not affect rates of crumbling or disappearance. On arid rangeland, however, dung decomposition was accelerated by nitrogen fertilization and irrigation (Lussenhop and others 1982). Mineral redistribution within a meadow shows up as short-term localized effects of dung pats and urine. The effects will be at a maximum one to two months after deposition and gradually decline over a period of about 18 months (Castle and MacDaid 1972).

Minerals are put into meadows through precipitation, fixation, mineral and organic matter decomposition, and runoff and sediments from surrounding slopes. Long-term effects of mineral redistribution from grazing likely become apparent only over many years. These effects show up as general changes in species composition, with increases in those animal and plant species best able to utilize the altered mineral state.

Mineral redistribution by people is potentially more of a problem than mineral redistribution by livestock. People redistribute minerals from distant ecosystems to meadows and their associated ecosystems. Livestock redistribute minerals within and among closely associated ecosystems.

An adult human will produce about 2.2 lb (1 kg) of dung and urine per day (Reeves 1979). Water makes up about 90 percent of the excrement; on a dry-weight basis, the daily production is 0.22 lb (100 g). From 1,000 visitor days, therefore, 220 lb (100 kg) of fecal material are deposited.

Decomposition rates of human dung in the Sierra Nevada are slow (Reeves 1979). The variables affecting decomposition rates of human and cow dung are basically the same, however. And decomposition rates of human dung may possibly be increased (at least on dry sites) by nitrogen fertilization, as was found effective on cattle dung (Lussenhop and others 1982).

Vegetation responses on and around small dung pits or "cat-holes" have not been studied. We cannot say, therefore, how human waste affects the vegetation of meadows. But meadows evidently do affect what happens to minerals and organisms in the dung and urine: meadows act as filters. Soils of the Sierra Nevada generally have a low degree of filterability (Reeves 1979). But soils of meadows were reported to be relatively efficient buffers against pollution of lakes and streams, possibly because of the organic matter content of meadow soils. Presence of a meadow between the place of fecal deposition and a lake or stream is therefore most desirable.

Burrowing

High rodent populations, particularly mice and pocket gophers, can considerably damage a meadow. The species composition may change through preferential grazing, and erosion may increase by channeling of water in burrows. Presence of rodents and their activities are not all bad, however. They are certainly a source of food for other animals (carnivores) and thus play a significant role in the functioning of meadow ecosystems. Also, their cultivation of the soil is often considered beneficial.

In 1965, high-elevation meadows in Sequoia National Park were much disturbed by rodents (Hubbard and others 1965). Most of the disturbance was caused by meadow mice (*Microtus* spp.); pocket gopher *(Thomomys monticola monticola)* activity was found only on a small scale. The mouse population was thought to have rapidly declined in late winter or early spring 1965 after reaching a high in 1964. It was thought that the population would prove cyclic, reaching a new high in 3 or 4 years. But followup study showed low mouse populations through 1969 (Giffen and others 1970).

Rodents in Gaylor Lake Basin, Yosemite National Park, were not distributed in either a random or uniform fashion (Klikoff 1965). Rodent activity was absent in wet meadows, but tended to be locally concentrated in dry meadows. Disturbance in moist meadows was intermediate. Belding ground squirrels *(Spermophilus beldingii beldingii)* and pocket gophers were causing most of the disturbance. The yellow-bellied marmot *(Marmota flaviventris sierrae)* was thought to cause less disturbance because it burrows under large rocks. Herbage consumption by the rodents was not considered a critical factor in plant distributions. But reduction of seed was thought to be a significant action of the rodents.

Pocket gophers in two meadows at Huntington Lake on the Sierra National Forest frequently cached food, used underground plant organs at all times, and used herbage mostly in winter by burrowing under the snow (Ingles 1952). Mound building began in May and reached a maximum in August and September. An estimated 7.5 tons (6.8 metric tons) of dirt for mounds were moved in one year on the two meadows. Gopher activities were, however, beneficial in building soil and conserving water (Ingles 1952).

"On wild land the burrowing rodent is one of the necessary factors in the system of natural well-being" (Grinnell 1923, p.149). Gophers aid soil formation and water conservation, and together with ground squirrels help to reverse the effects of soil compaction. The effects of pocket gophers on meadows and grasslands have received considerable study in other areas (Bronson and Payne 1958, Ellison 1946, Moore and Reid 1951). Through preferential grazing, gophers reduce some desirable forage plants. They also reduce some undesirable plants and thereby benefit some good forage plants. Gopher diggings make a relatively poor seedbed where the soil and vegetal cover are intact. But where plants are sparse, gopher diggings make a relatively good seedbed. They may provide a suitable environment for Sierra Nevada lodgepole pine *(Pinus murrayana)* seedlings in otherwise dense meadow vegetation. On meadows in good condition, gophers use a

relatively small portion of the vegetation and are not a serious problem. But on meadows in poor condition, a few gophers may use such a high proportion of plants that improvement is prevented. Most gopher activity is seen in those areas where livestock grazing has reduced the plant cover and where the soil displaced thereby is most liable to be exposed to erosion. Livestock grazing—at least overgrazing—appears to reduce the good, and to accentuate the bad, effects of gophers.

Rodents influence meadow decomposition by loosening the soil and mixing organic matter with it (Vilenskii 1957). They thereby improve conditions for decomposer activity and help to speed the recycling of nutrients.

Lodgepole Pine

Invasion of Meadows

Sierra lodgepole pine is listed as *Pinus contorta* var. *murrayana* (Krugman and Jenkinson 1974). It is also considered a separate species (Munz and Keck 1959). Common names for the species are Sierra lodgepole pine, lodgepole pine, and tamarack pine (Little 1979). Other varieties of lodgepole pine—*P. contorta* var. *contorta* and *P. contorta* var. *latifolia*—are in close kinship with Sierra lodgepole pine; available information on them should be considered. For convenience, this report uses *lodgepole* or *lodgepole pine* to refer to all varieties.

Lodgepole pine's remarkable physiological adaptability and its occurrence around and in mountain meadows make it a focus of concern to land managers. It has invaded many meadow sites. The trees reduce the area of open meadow, alter light and moisture available for herbaceous plants, and produce undesirable changes in species composition and productivity.

Invasion of meadow sites by lodgepole pine has been documented primarily by observations spanning many years. A sharp transition from open meadow to large, older trees gives the impression of a stable relationship between forest and meadow. A transition from open meadow to scattered small trees to larger trees to large mature trees suggests instability and change.

The best quantified data on invasion are provided by Vankat (1970). He determined ages of trees in and around meadows and dated the invasion by lodgepole pine as 1900 to 1905. Lodgepole pine reproduction was scarce between 1865 and 1900—the period of heavy grazing by sheep.

The following hypotheses related to invasion of meadows by lodgepole pine have been proposed (Leonard and others 1969):

> The establishment of lodgepole pine seedlings is inhibited or prevented at the germination stage by sod, dense meadow vegetation or dense organic surface material, and by saturated soil. Seedlings are inhibited from rooting by saturated soil and are highly susceptible to trampling on grazing sites free of exposed rocks. Growth and development beyond the vulnerable seedling stage are further inhibited by constantly saturated soil, and by trampling and browsing.
>
> A lodgepole pine that has developed beyond the seedling stage on a meadow site can be expected to follow a normal course of development and to grow relatively rapidly even in seasonally saturated soil, or constantly wet but well aerated soil. The development of a stagnant, constantly saturated root environment is one exceptional circumstance that can lead to the decline and death of a sapling or mature tree over a period of several years.

Assuming that these hypotheses are true, in order to stop or reverse the invasion process, we need to understand the requirements of lodgepole pine for seed germination and seedling establishment. And we need to understand how management can modify the meadow environment to prevent occurrence of those requirements.

Growth and Establishment

Lodgepole pine can become established, grow well, and reproduce in soil environments that apparently are unfavorable to other conifers. It also grows under conditions in which the other conifers grow, including near tree-line environments. Although not always the dominant tree species, lodgepole pine probably grows in most montane and subalpine meadow edges of the Sierra Nevada.

Sierra lodgepole pine grows in an elevational range of about 5,000 to about 11,000 ft (1,525 to 3,353 m). the trees grow from 50 to 130 ft (15 to 40 m) tall. They produce pollen in May and June (Krugman and Jenkinson 1974), and cones ripen and disperse seed in September and October. Sierra lodgepole pine does not have serotinous cones; rather, the cones open soon after maturity and do not persist long on the trees. Seed production can start at 4 to 8 years of age, and normally a large seed crop is produced each year. The average number of cleaned seeds is 117,000 per lb (257,938/kg). About 200 lb (91 kg) of cones are required to obtain a pound of seed. Stratification of fresh seed is not needed to induce germination; however, for stored seed, stratification at 33° to 41° F (0.6° to 5° C) in a moist medium is recommended. Germinative capacity averages about 75 percent. A 70 percent germination of lodgepole pine seed collected at Rock Creek in Sequoia National Park was reported (Harkin and Schultz 1967).

Lodgepole pine regenerates best on mineral soil or disturbed duff free of competing vegetation and in full sunlight (Tackle 1961). The seed of lodgepole pine requires a small amount of light for germination, but the seedlings require considerable light to ensure satisfactory reproduction (Mason 1915). The best seedbed is a mineral soil with plenty of heat and available moisture; layers of needles or undecomposed humus may dry out before roots can reach the mineral soil. In Oregon, seedbeds of mineral soil supported about twice the number of seedlings as undisturbed litter (Trappe 1959). In summary, lodgepole pine needs a mineral seedbed in well-lit, warm, moist environments.

One study reported on effects of low temperatures and compared lodgepole and ponderosa pine seedlings (Cochran

and Berntsen 1973). Thirty-six-day-old seedlings suffered greater mortality from night temperatures of 18° F (–7.8° C) than did 22-day-old seedlings. Younger seedlings of lodgepole pine can therefore stand lower night temperatures in spring than older seedlings. If a period of warm temperatures (just above freezing) is followed by temperatures of 20° F (–6.7° C) or less, mortality is reduced. Also, seedlings grown until fall under natural photoperiods can withstand minimum night temperatures of 15° F (–9.4° C), showing that shortening photoperiods are conducive to hardiness. It appears that only extremely cold night temperatures in spring could prevent lodgepole pine establishment in meadows. Such temperatures may occur in some meadows because of cold air drainage.

If other conditions are suitable, abundant germination of lodgepole pine seed may be expected in meadows soon after soil warms in spring. A study of the effects of temperature on germination of lodgepole pine seeds found that the optimum temperature was about 70° F (21.1° C) (Bates 1930). Temperatures above 82° F (27.8° C) or below 60° F (15.6° C) reduced germination. Fluctuating temperatures with the daily mean at 65° to 70° F (18.3° to 21.1° C) gave the best rates of germination. The response to fluctuating temperatures was considered to be linked with the species' habit of reproducing in exposed areas with great daily extremes in temperatures. The maximum and minimum temperatues of 48° and 38° F (8.9° and 3.3° C) at one centimeter from the south side of rocks (Leonard and others 1968) approximate the fluctuation stated earlier.

At 50 percent full light intensity, lodgepole pine seed germination was 22 percent (Boerker 1916). At 16 percent of full light intensity, germination was 7.5 percent; at 2 percent of full light, 3.5 percent. These data indicate lodgepole pine's need for light.

Moisture Effects—Physiological responses of lodgepole pine to soil moisture were studied along Rock Creek in Sequoia National Park (Leonard and others 1968, 1969). Pressure chamber readings on cut branches were taken as measures of physiological response. A branch (with its cut end protruding through a stopper) was sealed in a pressure chamber. The chamber was gradually pressurized with dry nitrogen gas; the pressure was recorded when water became visible at the cut surface.

Except on a porous moraine (dry site) in 1968, lodgepole pines under all site and seasonal conditions registered pressures almost entirely below 200 pounds per square inch (psi) (14/kg/cm^2). Diurnal pressures in September, however, peaked at about 225 psi (16 kg/cm^2). Pressures on the dry site approached 250 psi (18 kg/cm^2) toward the end of summer. A pressure of 200 psi (14 kg/cm^2) represents a level of stress of significance for lodgepole pine. That pressure was exceeded only under circumstances of depleted soil moisture, excessively high evaporative stress, fungal infection of the needles, or an obviously poor crown condition.

Midwinter pressure readings showed that soil moisture was readily available. Except for a few surface centimeters, the soil was not frozen. Water and nutrients are therefore available for metabolic processes of lodgepole pine even in the dead of winter. Water movement in the trees is normal, at least when leaf tissue is not frozen.

Lodgepole pine is tolerant of soil saturation, thus able to survive and grow in meadows. No differences in physiological response were found among trees along a gradient from saturated meadow soils to porous moraine soils. Soil saturation did not appear detrimental to lodgepole pine. Deliberately flooded trees had low pressures and high rates of transpiration, indicating efficient water absorption by the flooded roots.

Continuous saturation lasting for several years could nevertheless contribute to tree mortality. Mortality of lodgepole pine in some normally saturated situations was observed. Long-term saturation of the trees may result in preferential attack by fungal infections of the needles. Characteristic fungal reproductive structures in needle specimens indicated that damage was caused by *Hypodermella montana*. Crowns of weakened trees were sparser than uninfected trees and showed some yellowing. Severely affected branchlets had shed all but the current crop of needles and their terminal buds were often dry and brown. Although such fungal infections may not kill the trees, they can weaken them. These trees may then become susceptible to secondary attack by bark beetles. Evidence of bark beetle attack was observed in dead trees.

Disturbance by Livestock and Rodents—Grazing disturbance by livestock is the usual reason given for the invasion of meadows by lodgepole pine. Grazing—especially sheep grazing—and its influences on the vegetation and soil modified the meadow environment creating a suitable niche for lodgepole pine (Vankat 1970), and I believe that statement to be true. However, I also believe the proposition must be accepted that meadow invasion by lodgepole pine can be autogenic—the result of plant succession—and not always the result of allogenic effects.

On biologically and geologically stable meadows, the opportunities for invasion by lodgepole pine are few (Benedict 1981, 1982). Instability can result both from natural causes and from human activities.

If geologic or climatic change can induce instability in a meadow system by permitting lodgepole pine invasion, geologic or climatic change can induce instability in a forest system by permitting meadow invasion. That such events occur in the Sierra Nevada has been well demonstrated (Wood 1975). Over geologic time, meadows have developed, disappeared, and redeveloped at the same geographical locations.

From studies of stand age structure of trees contiguous with meadows, it has been hypothesized that sheep grazing and fires set by sheepherders to facilitate the movement of their flocks prevented regeneration of lodgepole pine, thereby stopping the process of invasion (Vankat 1970). This situation was photographed at Jackass Meadow on what is now the Minarets Ranger District of the Sierra National Forest (Sudworth 1900). The picture shows a fence line situation. The caption states that within the fenced area lodgepole pine reproduction was abundant, but outside, where sheep grazed

heavily, only scattered large trees grew but did not reproduce. Grazing pressure and recurrent fires set by sheepherders were responsible for the lack of tree regeneration in the Rock Creek area of Sequoia National Park (Sudworth 1900).

Certainly, we can associate overgazing with soil disturbance and loss of vegetative cover. Sheep grazing during one 9-year period (1864-1873) was blamed for creating "a gray sea of rolling granite ridges, darkened at intervals by forest, but no longer velveted with meadows and upland grasses" on the Kern Plateau (King 1902, p. 351). Sheep, through trampling and grazing, can destroy fragile sods and effectively denude some meadows.

Although it prevented or checked the invasion of meadows by lodgepole pine, heavy grazing may have provided bare mineral soil and reduced mulch and competition, thereby predisposing meadows to invasion. We know that large areas of meadow were invaded by lodgepole pine soon after 1900, and that this invasion coincides with the end of heavy concentrations of sheep. We can therefore hypothesize that human activities, within historic times, have both impeded and encouraged lodgepole pine invasion.

Probably, lodgepole pine invasion of meadows is most strongly related to sheep grazing; however, cattle can cause severe disturbance to meadows. That continued invasion by lodgepole pine into meadows can be ascribed to forces set in motion during the time of sheep grazing is difficult to determine. We know that the current invasion has been occurring on ungrazed meadows and on meadows grazed continually since about 1900.

Grazing may indirectly cause invasion by lodgepole pine by affecting gopher activity and populations of some seed-eating rodents. Overgazing often aggravates the normal effects of gophers. And overgrazing may expose other rodents to greater predation, thereby reducing the amount of lodgepole pine seed consumed and improving changes for seedling establishment.

Although he did not fully evaluate the effects of pocket gophers, Buchanan (1972) suggests that they may play a role in lodgepole pine invasion. Surface soil deposits may benefit survival of lodgepole pine seedlings. These deposits provide the mineral seedbeds apparently required. Gopher mounds are slightly raised, not as deeply shaded by herbaceous vegetation as the undisturbed soil surface, and are relatively free of competing vegetation. Pocket gopher mounds covered about 5 percent of the meadow and 4 percent of the meadow-forest ecotone, providing a sizable area of suitable seedbed (Buchanan 1972). Where livestock no longer graze—as well as where they do—gopher activities may contribute to meadow invasion by lodgepole pine.

Snow Depth—In the Sierra Nevada, most lodgepole pine seedlings become established in years of low snowpacks (Wood 1975). Length of the snow-free period may be the most critical variable in tree invasion of subalpine meadows (Franklin and others 1971). Tree invasion is related to periods of below-normal snowpacks and earlier snowmelt or to a long snow-free period after melt. A good seed crop followed by an early melt of snow can be expected to result in significant tree establishment. Where snow is deep, conifers do not usually reproduce (Billings and Bliss 1959). In the "ribbon forests" of the Rocky Mountains, tree seedlings were protected by snowpacks on their windward side (Billings 1969). Deep snowpacks resulted from the ribbon of trees immediately to windward. Adjacent to and in the lee of that ribbon, snowpacks were generally too deep and melted too late to permit tree establishment.

From these reports, we may expect that deep, long-lasting snowpacks tend to inhibit lodgepole pine invasion of meadows. But studies have not proven this to be fact.

A weather modification that increases snowpack could enhance conditions for lodgepole invasion (Buchanan 1972). Lodgepole pine seedlings, 2-0 stock, were planted in meadows, forests, and forest-meadow ecotones of the Bridger Range, Montana. Survival of the seedlings was lowest in the meadows and increased with increasing snow depth up to 7.9 ft (240 cm). Naturally occurring lodgepole, 1.7 to 6.7 ft (0.5 to 2 m) tall, increased in frequency with increasing snow depth ($r = 0.7$). Frequency of small seedlings was significantly correlated ($r = 0.36$) with snow depth on the meadows. Survival of planted lodgepole pine and presumably natural reproduction was explained by the relationship of snow and availability of late-season moisture. From this, we may expect the greatest invasion activity at the forest-meadow ecotones where snow accumulates to depths of not over 8 ft (244 cm) and melt is slow.

It seems reasonable that periods of low snowpack and early melt may be necessary for seedling establishment. Once established, the young trees are protected from cold temperatures, wind, physiological drought, and browsing by a deep, long-lasting blanket of snow. If the snowpack is deep enough and lasts long enough, survival of invading lodgepole pines should be reduced.

In the Sierra Nevada, lodgepole pine is frequently cut, piled, and burned to enlarge or reclaim a meadow. Although cut for lumber, it is not usually considered prime timber for manufacture. It is cut commercially for fence posts and firewood. Size and orientation of the openings from harvest or other removal of lodgepole pine influence snowpack depth and melt. General references on forest-opening effects on snow accumulation and melt are available (Anderson 1956, 1967, Berndt 1965, Gary 1980, Niederhof and Dunford 1942).

The largest snowpacks are observed on southern edges of openings shaded by trees, a few meters immediately to the lee of bordering trees, or where opening width is about equal to the height of the trees around it. Snowpacks on large forest openings disappear before the pack in long, narrow openings with an east-west orientation. Because of reduced radiation, the pack melts more slowly in openings with a north aspect and on well-shaded openings. Northern edges of openings usually accumulate less snow because of direct solar radiation and back radiation from trees. Where meadow reclamation is an objective, lodgepole pine harvest should be planned to take full advantage of these relationships.

Soil Conditions—The nature of the soil may influence whether lodgepole pine is able to invade a meadow. Lodge-

pole pine grows in shallow depressions (Howell 1931). There, clay pans form resulting in a perched water table. Lodgepole has been observed growing on two general soil types—well and excessively well-drained, moderately coarse to coarse-textured soils; and poorly and very poorly drained, organic soils (Stephens 1966). Medium to moderately fine textured soils, and moderately well drained, but moderately coarse to coarse textured soils lacked lodgepole pine. Lodgepole pine is considered a pioneer species on droughty, low fertility soils and on wet, organic soils (Stephens 1966). It is also considered a pioneer on bars of cobbles, rocks, and sand along the Kern River (Hubbard and others 1966).

A relationship of seedling, sapling, and larger trees with rocks in meadows has been observed (Leonard and others 1968). Higher maximum soil temperatures immediately adjacent to rocks and absorption of energy by the rocks could provide a more favorable environment for lodgepole pine than meadows away from rocks. Other microenvironmental conditions near rocks that may favor lodgepole establishment include exposed mineral soil, better soil texture, better aeration and drainage, and early snowmelt.

Soil moisture near rocks was 143.3 percent. But 4 ft (1.2 m) away from rocks, soil moisture was 351.3 percent. Soil particles 2 mm or less averaged 83.5 percent of the bulk sample near rocks and 96.3 percent of the bulk sample four feet (1.2 m) away from rocks (Giffen and others 1970). Moisture content is not independent of the percentage of fine soil particles.

Lodgepole pine roots have an affinity for rocks (Preston 1942). Roots tend to group adjacent to a large rock and some roots may penetrate the rocks. This affinity may be a nutrient reponse in an otherwise rather sterile soil.

In my studies of mountain meadows, I observed an apparent affinity of lodgepole pine for rocks in some, but not all, meadows. Where the soils are mainly of organic origin, rocks may provide a better habitat. On mineral soils, texture and drainage may be the controlling characteristics.

Fire

Fire as a part of the natural environment of many vegetation types in California has been receiving greater recognition during the last several years (Botti and Nichols 1980, Gordon 1967). Evidence strongly suggests that fire plays a significant role in the evolution and maintenance of meadows of the Sierra Nevada (Sharsmith 1959). Fire has been thought to influence the forest-meadow boundary (DeBenedetti and Parsons 1979, 1984; Parsons 1981).

Fire has a role in meadow evolution and maintenance in other mountain regions as well. In the Cascades of Washington and Oregon charcoal and rotting wood are uncommon in meadow soils and fire is not an important variable (Franklin and others 1971). In the Olympic Mountains of Washington, however, fire is a major variable in meadow creation and maintenance (Kuramoto and Bliss 1970). Between 1870 and 1880 fire destroyed much of the subalpine forest around James Peak, Colorado, permitting downward extension of alpine vegetation (Cox 1933). Destruction of ribbon forest by fire could result in new snowdrift patterns and replacement of forest by alpine or subalpine meadows in the Rocky Mountains (Billings 1969). And old records show fire is a natural ecosystem component of "Crex Meadows" in northwestern Wisconsin (Vogl 1964).

Meadows are not likely to burn when the herbage is grazed or in years of normal precipitation. Meadows may burn when herbage is tall and dry and during droughts. A fire crossing a meadow having high soil moisture usually consumes only current growth and some mulch. With very dry conditions, fire may damage the meadow considerably (DeBenedetti and Parsons 1979, Parsons 1981). To understand how fires affect meadows, we need to know how fire frequency and fire incidence relates to meadows and their watershed.

Fire frequency is the time between fires at particular points; fire incidence is the time between fires within a particular area (Kilgore and Taylor 1979). The larger the area considered, the greater the fire incidence. A higher fire incidence may be expected on its surrounding watershed than on a particular meadow. And, with their normally drier fuels, points in forests may be expected to burn more frequently than points in adjacent meadows. Fires in their watersheds are therefore apt to influence meadow ecology more often than fires directly on meadows. Increased water flows and sedimentation are likely the most significant effects of fires in the watershed. Fires that directly burn meadows will, of course, have greater and immediate effects on meadow ecology.

Fire History

Brush and forest fires have prevailed in California since the earliest recorded times (Sterling 1904). Fire frequency in portions of Kings Canyon National Park and Sequoia National Forest was 8 to 18 years between 1478 and 1875 (Kilgore and Taylor 1979). Fire incidence in the watershed areas was 1.7 years during that period.

Particular forest sites burned more often than expected from lightning alone (Kilgore and Taylor 1979). Burning by American Indians also augmented fire frequency. After 1875, fire frequency decreased markedly. Only 5 percent of all fire scars dating back to 1478 dated betwen 1875 and 1939. During 10 years (1865 to 1875), burning by herdsmen and others replaced burning by American Indians (Kilgore and Taylor 1979). American Indians regularly burned the mountains and sheepmen came and continued the practice (Sterling 1904). Setting fires apparently was declining as a regular practice even before fire suppression became effective.

Fuel conditions at higher elevations are lighter and fire crowning is currently not as serious a problem as at lower elevations (Kilgore 1971). Fire scars at the higher elevations were less frequent than at the lower elevations (Kilgore and Taylor 1979). Above about 7,500 ft (2.286 m), any role played by fire in meadow development diminished in importance (Sharsmith 1959). Lower elevation meadows are likely more often affected by forest fires than are higher elevation meadows.

Major fires (or at least fires of more than a few acres) may be required to produce geologic effects on meadows. Evidence of fire is frequently found in profiles of meadow soils. Five layers of charcoal were deposited in one meadow (Wood 1975). The deposits were all less than 1200 years old. These layers support that idea that major fires in the watersheds normally occur at intervals of 250 to 300 years.

Fire associated with the history of meadows along Rock Creek in Sequoia National Park was on approximately that time scale (Leonard and others 1969). Charcoal was found in the 7 to 15 inch (18 to 38 cm), 21 to 24 inch (53 to 61 cm), and 29 to 33 inch (74 to 84 cm) layers of the soil profile. The charcoal was found mainly in sand and gravel sediments, indicating that it came from slopes above the meadow. Evidence of the most recent fire was found on the slopes, and older trees seemed to correspond to earlier charcoal deposits. Ages of the trees dated the last two major fires in the Rock Creek area at 1900 and between 1700 to 1670.

Because of its relatively thin bark, lodgepole pine is more susceptible to fire damage than other pines (Mason 1915). Fire is most destructible to dense young stands. Higher mortality of lodgepole pine than of red fir *(Abies magnifica)* due to fire has been reported (Kilgore 1971). Although many mature lodgepole pines were killed, burning appeared to provide conditions favorable to lodgepole pine seedling establishment.

Evidence of two fires along Rock Creek was found (Harkin and Schultz 1967). The oldest was judged to be about 70 years old and would correspond with the 1900 fire cited to earlier. A younger, 1962 fire was thought to have been caused by a fisherman. That fire spot burned over a beaked sedge site and a hanging meadow that contained willows (*Salix* spp.). Upon leaving the meadow, the fire burned upslope and killed most trees on drier areas. Trees and willows within the meadow were not killed, but patches and stringers of trees outside the meadow boundaries were killed (Giffen and others 1970).

Part of a meadow along Rock Creek was intentionally burned in late fall 1969 (Giffen and others 1970). The burn was reportedly light because of low fuel volumes. Small trees in the path of the fire were not killed. In 1970, no visible differences between burned and unburned portions of the meadow were evident.

Prescribed burns to improve deer habitat on the Sierra National Forest have affected some meadows (Bertram 1982). Although not actually crossing the meadow, fire destroyed some down logs in Three Springs Meadow. Destruction of these natural check dams could have resulted in accelerated erosion and gullying. At Cabin Meadow, some areas had become dry enough to allow white fir *(Abies concolor)* to establish. Dead lodgepole pines in those areas were felled to provide ground fuel. The burn produced substantial kills of lodgepole pine trees up to sawlog-size and of sapling-size white fir. Meadow vegetation in the burned areas had expanded by the next year.

The only documented instance of a wildfire burning a meadow in the Sierra Nevada is that reported by DeBenedetti and Parsons (1979, 1984) and Parsons (1981). Started by a lightning storm, the fire reached Ellis Meadow, Kings Canyon National Park, at 8,800 ft (2,682 m) in early August 1977. By the end of September, the meadow was still burning and about 60 percent of its 30 acres (12 ha) had been burned. Intensity of burning and depth of burning varied from place-to-place in the meadow. Areas with wideleaf sedges burned most intensively, and the ash layer reached a depth of 15 inches (38 cm) in some places. One year after the fire, grass and grasslike species made up only 8 percent of the ground cover; by 1981, they had increased to about 75 percent. The most serious aspect of this fire appears to be the loss of the deep peat layers built up over many, many years. The species composition reported the year after was mostly what one could expect to find on heavily grazed meadows. In 1981, vegetation was thought to be succeeding toward species characteristic of the prefire state. Catastrophic change had not occurred.

Fire Management

Not enough is known about fires in meadows to prescribe their use as a management tool. We have much to learn about the use of fire in the meadow environment, and it must be used with proper safeguards. A few observations are available, however.

A light fire across a meadow appears to affect vegetative composition only slightly. But changes in herbaceous vegetation that would result after a hot fire in a meadow appear undesirable. Because of this, any prescription for the use of fire must consider the soil moisture condition.

Because it is unlikely that fire crosses a meadow until the vegetation is mature and dry, fire affects the forage available only slightly. On grazing allotments, however, a season of nonuse may be needed to provide enough herbage to carry a fire.

Fire on a watershed can have marked effects. Probable changes in runoff and sediment loads need to be considered-- especially if a large area is to be burned. A large increase in runoff through a meadow may result in gully formation. But a sediment load in the runoff is necessary if gullies are to be filled in behind check dams. Logs in meadows are natural check dams, and care should be taken not to burn them.

Years of low snowpack and early snowmelt may favor lodgepole pine seedling establishment. Unless followed by late spring and summer rains, these conditions are also likely to be conducive to fire. Fire-caused mortality of lodgepole pine has been reported, but fire has also been reported to stimulate lodgepole seedling establishment.

It appears possible to use fire to manipulate succession on the edges of meadows. Succession to forest was set back at Cabin Meadow on the Sierra National Forest (Bertram 1982). Tree invasion at Ellis Meadow in Kings Canyon National Park had been slowed by fire and no trees were established in burned areas through 1981 (DeBenedetti and Parsons 1979, 1984; Parsons 1981). Trees growing in hummocks in the meadow, however, survived the fire. Light fires may kill some seedlings. Fires hot enough to kill trees established in the meadow proper will likely do great damage to the meadow

itself. And under conditions suitable for a hot fire, restriction of the fire to the target area may be almost impossible.

Gully Erosion

Prevention or control of erosion is the key to maintaining or restoring the hydrologic characteristics of a meadow, thereby maintaining or restoring the meadow itself. Erosion removes the protective sod and productive topsoil, and subsequent gully formation alters meadow hydrology by lowering the water table. Poorer soil and altered hydrology induce change in the vegetation—usually toward a less desirable plant community.

Erosion control in mountain meadows has a threefold purpose: to check the progress of active gullies, to bring about a refilling of the deeper gullies, and to restore the water tables. The objective is not only control, but rehabilitation.

Erosion is the transporting of soil by the actions of wind or water. It is a natural geologic process whereby disintegrated rock and soil are picked up and, if transported by water, moved to and deposited at a lower elevation. Sheet erosion is the removal of layers of soil from continuous areas. In mountain meadows, the natural sod of sedges, grasses, and forbs usually protects the soil from sheet erosion, but it does not always offer adequate protection against gullying. Gully erosion is the cutting of channels, and on-site erosion in meadows is most apparent as gullies.

A gully characteristically works up a stream from the lower end. A common example is where a main stream bed is quickly cut down by abnormal erosion, leaving undisturbed tributary channels perched at the former level. Such tributary channels cut back rapidly and in mountain meadows often form deep gullies where only shallow swales existed before. Where a break in the channel gradient occurs, a waterfall is formed. Breaks in the channel gradient designate a gully as a discontinuous type (Heede 1960). The falling water churns against the bare earth behind the falls and erodes the soil rapidly at this point. This is called "plunge-pool action" (Emmett 1968). An alternate erosive action is "sapping." Groundwater flows from the face of the gully head, washing away friable material below the sod. At the top of the gully head, the soil—held together by the roots of the meadow sod—is able to resist erosion. As undercutting proceeds by plunge-pool action, sapping, or both, the sod-crest begins to overhang the subsoil. From time to time, chunks of this overhanging sod break off and fall into the gully. The rate a gully head works upstream therefore depends on the soil structure and texture, the amount of root binding material, the maximum flow of the gully, the flow of groundwater, and the gradient of the channel. Its progress is steady until measures are taken to stop the undercutting. As erosion of the gully heads proceeds, individual segments of a discontinuous gully join to form a continuous gully (Heede 1960).

Most meadow gullies have vertical sides with rims of overhanging sod. Unless corrected, this condition continues by constant undercutting of the sides. This situation should not be confused with the overhanging sod along permanent streams. There, the overhang is considered beneficial as fish habitat.

Erosion Control

Gully control is an effort to restore a stable condition in an intrinsically unstable situation where natural forces are seeking to establish a new stability. On the other hand, it lowers the effects of extrinsic variables to below their threshold levels in order to achieve stability.

The concept of meadow stability has been discussed (Benedict 1981, 1982, and Benedict and Major 1982). A meadow may be geologically and biologically stable, geologically stable and biologically unstable, or geologically and biologically unstable. The situation of a geologically unstable and biologically stable medow is untenable. As long as the geologic conditions favorable for meadow formation persist, biological instability can eventually be overcome. Biologic stability has to do with condition trend rather than the ability of a meadow to withstand and recover from abuse.

Geologic stability depends on favorable conditions for meadow formation and maintenance. As such, the concept of geologic stability includes stability in intrinsic and extrinsic factors.

Presence of a dissecting gully has been taken as evidence of a meadow's instability (Wood 1975). Wood attributed meadow instability largely to damage to the sod by livestock grazing. Under the protection of undamaged sod, most gullied meadows had previously built up steeper slopes than allowed by their watershed areas.

And that is the concept of geomorphic thresholds (Schumm 1977). As material from the watershed is deposited, the valley (meadow) slope progressively increases until a threshold slope is reached and erosion of the valley (meadow) fill occurs. "Large infrequent storms can be erosionally significant, but only when a geomorphic threshold has been exceeded are major permanent changes the result" (Schumm 1977, p. 81). Variables intrinsic to landform development can induce erosion, and a change in an extrinsic variable is not necessary for the geomorphic threshold to be exceeded. On the basis of geomorphic threshold, then, livestock grazing or other disturbances should result in significant gully formation only after a meadow has evolved a slope-to-watershed relationship at which erosion, rather than deposition, can occur. But once slope is at or near threshold, the intrinsic process of erosion may be accelerated by extrinsic variables (Wood 1975).

Extrinsic variables also have thresholds (Schumm 1977). Grazing by livestock, for example, is an extrinsic variable. If the grazing effect is above the threshold level, the density of the vegetative cover may be reduced. In turn, though volume is constant, the hydrologic threshold of flow velocity may be exceeded because less vegetation is available to dissipate the energy. Soil erosion and possibly gully formation are the result

Other extrinsic variables which may cause intrinsic variables to exceed their threshold values include denudation of the surrounding hills by logging, fire, or both; construction of highways and roads where roadway drainage concentrates

water flow into a meadow; and destruction of the protective sod for crop agriculture.

Control Measures

There are no recent guides for erosion control specific to meadows of the Sierra Nevada. Instructive, up-to-date procedures for erosion control are, however, available (Heede 1965, 1966, 1977). And his discussion of early erosion control structures is excellent (Heede 1960).

The handbook on erosion control in mountain meadows (Kraebel and Pillsbury 1934) is directly applicable to meadows of the Sierra Nevada. It covers materials, design, and construction of control structures that are useful where costs or esthetics limit machinery, such as in wilderness areas. Unfortunately, the handbook is not generally available. But, as appropriate, parts of it are summarized below.

A complete control project includes an initial assessment, proper range management, rodent control, building of proper structures, and planting of appropriate vegetation. The initial assessment aims to determine if the meadow slope is over the geomorphic threshold, if erosion is occurring at an accelerated rate, and if accelerated erosion results from human activities on the meadow or other activities in its watershed. Proper range management aims to restrict grazing until the gully has been stabilized, followed by a regime that maintains the stabilized condition. Elimination of grazing is not called for if the meadow is geologically and biologically stable. Proper grazing management can improve meadow condition in a situation of geologic stability but biologic instability. Elimination of grazing may slow the demise of a meadow but, in a situation of geologic instability, will not restore a meadow. Rodent control aims to prevent excessive burrowing and consumption by rodents of young plants artificially introduced as part of the gully control. Building proper structures is designed to fill gullies and control erosion until natural or introduced vegetation becomes vigorous enough to make control permanent. The focus must be on the watershed mouth (Heede 1960, 1981), representing the base level of the watershed: what happens there affects the entire upstream area. The key is to first control that point. Work can than proceed upstream. Planting appropriate vegetation makes control measures more effective and permanent.

Materials Specification—The criterion for selecting materials for gully control structures is that satisfactory materials be readily and cheaply available.

Dam brush—chaparral is a satisfactory material, although tree branches are easier to handle. The brush should be somewhat flexible, preferably green, and heavily leaved. Dry, brittle brush is difficult to handle and does not make a satisfactory dam. The brush should be cut small enough to pile easily into a dense mass. The maximum convenient length is 3 or 4 ft (0.9 or 1.2 m). **Apron brush**—the requirements for apron brush are the same as for dam brush, except that it must be long and flexible. Willow and branches of various evergreen trees are excellent. **Litter**—any finely textured vegetative material can be used beneath aprons and against the upstream faces of dams. The best and most readily available material is the forest litter or leaf mulch that may be raked up from beneath the trees. **Trees**—where specified, trees should be freshly cut and have a dense foliage. Only evergreen trees should be used. **Logs**—these should be straight, sound, and of the sizes specified. **Posts**—any good fence post material is satisfactory. **Stakes**—any sound coniferous wood that does not rot quickly is recommended. If possible, use willow stakes, which may take root and grow. **Poles**—these should be straight and of sound wood. **Rock**—flat or angular rocks are far superior to round cobbles, since they have less tendency to roll during floods. Never take rocks from gully bottoms or other places where rocks provide a natural pavement. **Poultry netting**—commonly called "chicken wire," any convenient width is suitable. **Wire**—galvanized iron wire (No. 9 to No. 12) is recommended.

Gully Head Control—Plugs or mattresses constructed against a gully head have the immediate effect of stopping the plunge-pool action in underminimg the head. They may not be effective against sapping action, however. A compact layer of litter must be packed against the head itself to stop undercutting, against the side near the head to prevent side cutting, and along the bottom to serve as cushioning apron. The litter must be firmly held in place to prevent being washed out during floods. This usually entails placing good-sized tree branches or brush over the fine material which, in turn, must be secured with posts, stakes, rocks, or other large material. The catch basin behind a check dam placed a few yards or meters downstream from the gully head will fill with sediment within a few years. This reduces the height of any drop in the stream bed level between the dam and the gully head and minimizes the danger of a new gully starting.

If the channel carrying runoff immediately above the gully head is heavily sodded, and if, as a consequence, a definite overhang of the crest is at least 6 inches (15 cm), it is practical to plug the head without modifying the slope. If the gully head has no definite overhang and slopes slightly upstream, it is advisable to cut the slope back so that runoff has a gradual, rather than vertical, drop. To protect this bare slope until the catch basin behind the downstream dam has become silted, the headslope must be covered with a mattress that extends along the bottom of the gully for 4 ft (1.2 m) or more to serve as an apron. Where gully head plugs or mattresses are not feasible, gully heads should be sloped back to the natural angle of repose of the soil and permanently stabilized with appropriate vegetation.

Flumes have sometimes carried water over a gully head. But the expense and care of their construction is ordinarily prohibitive. Spreading water—diverting it away from the gully above the head by a system of ditches—is preferable. Unfortunately, sites where water spreading is feasible are uncommon in the Sierra Nevada. In some situations, however, meadow-edge trees can be felled above gully heads to effectively spread the water. Trees falling into meadows is a natural process as evidenced by the presence of logs in the profiles of most montane meadows in the Sierra Nevada. And, frequently, trees falling across drainages will result in the development of a kind of hanging meadow.

Gully Channel or Bank Control—Mountain meadow channel control requires stabilizing the banks of well-established gullies and stream channels that traverse or border the meadows. Meandering streams that need control with jetties, riprap, and similar structures are seldom, if ever, observed in mountain meadows.

Channel control consists of breaking the current, filling washes, and bank fixation. Light structures, such as hog wire or chicken wire fences, can be constructed to break the force of water. Such structures, where effective, result in the deposition of materials suspended in the water. Logging slash or similar material—tree tops, branches, and trash from cutting posts—can be placed in the channels. Such materials tend to slow the current and cause silt to be deposited. Bank fixation requires that banks of gullies and streams be sloped to the natural angle of repose of the soil. The aim is to make the bank sufficiently stable to support a cover of either natural or introduced vegetation. Normally, banks of about 70 percent slope are stable, but some soils may be stable at steeper angles. Most meadow gully banks become naturally overgrown when sloped back. If revegetation is required, sod should be planted or the slopes seeded. If the channel banks become loose and erode during rain, wattles should be constructed to stabilize them.

Once cut back, the slope usually requires protection until vegetation becomes fully established. A system of contour wattle construction, as on road slopes, can be adapted to control gully banks. A wattle is a continuous bundle of vegetation intended to hold the soil and interrupt the water flow down the slope. Wattles are usually spaced every 2 to 4 ft (0.6 to 1.2 m) along a slope. They can be effective on banks that have been sloped, where side gullies have started or are apt to start, on banks or gully heads that have a loose soil and little vegetative cover and are sloped to the natural angle of repose of the soil, and on wide or deep gully heads too large for practical use of normal gully head controls.

Several types of wattles are used. Sod strips 10 to 12 inches (25 to 30 cm) wide should be spaced about 30 inches (76 cm) apart. Terraces need not be made. The strips must be bedded firmly into the slope. Continuous hay or pine needle rope wattles are constructed by filling small trenches cut around the slopes on the contour or by using willow stakes to hold the ropes in place. Similar wattles can be constructed of brush.

The method of wattle construction depends on soil type, looseness of the slope, soil moisture conditions throughout the year, and anticipated flood flow. Normally, a combination of wattle types provides the best slope protection at the least cost.

On a soil moist enough all year to grow a heavy sod, the following wattles are recommended:

- Where the soil is not subject to rapid erosion, use sod strip wattles only. This condition frequently exists along a main channel through a meadow where the banks are slowly receding and broadening the stream bed. The sod eventually covers the entire slope and protects against floods along the channel parallel to the wattles.
- Where the soil is not subjected to a heavy flow, but erodes easily, alternate sod strip with staked rope wattles. Usually, staked rope wattles can be spaced 4 to 5 ft (1.2 to 1.5 m) apart, with sod strip wattles planted midway between.
- Where erosion is relatively rapid and the soil is light, alternate sod strip with brush wattles. Brush wattles can be spaced 3 to 5 ft (0.9 to 1.5 m) apart with sod strip wattles midway between.

On a soil too dry to grow a good sod, the following wattles are recommended:

- Where erosion is slow, space trenched rope wattles 30 to 40 inches (0.8 to 1.0 m) apart.
- Where erosion is expected to be faster, alternate trenched and staked rope wattles.
- Where conditions are most severe, use staked brush wattles.

Check Dams—Reclaiming a gully with check dams stops headward erosion (especially when check dams are combined with head controls), stops deepening and widening of the gully, and stops or minimizes formation of side gullies. The area of the gully itself is reclaimed and the water table is raised.

For the purpose of erosion control, check dams in gullies decrease the velocity of the water down the gully. By decreasing velocity, silt is deposited in the gully. With enough properly designed check dams, the gully stops eroding and becomes filled with the deposited material. For the gully to fill, however, erosion must be taking place somewhere above the point of deposition.

Because of poor foundation conditions generally found in meadow gullies, check dams in mountain meadows should be keyed into the channel bottom and channel banks and should be able to carry the maximum expected flow (Heede 1960). Numerous low dams along a gully are preferable to a few high dams. Low check dams are 3 to 4 ft (0.9 to 1.2 m) high. Low check dams do not usually wash out; but if they do, less flood damage results. A series of low dams should first be constructed along the gully. When the catch basins behind these dams have filled, another series of dams can be built on top of or just upstream of the original dams. Check dams should be semipervious rather than impervious. However, erosion may start again when check dams made of brush, forest litter, and other such materials rot away.

Proper spacing between dams depends upon the gradient of the gully. The minimum interval used should make the crest of one dam level with the apron of the one above. Heede (1960) refers to this as the "head-to-toe rule." But this may be an inefficient rule, since deposits behind a check dam possess a gradient. A more proper interval positions the apron of the next higher dam at the highest point expected to be reached by deposits behind the lower dam. The criterion is the gradient of the deposits, some ranging from 5 to 6.5 percent (Heede 1960).

Dams are more economically built, more effective, and more stable if placed in key locations (Heede 1960). Continuous and discontinuous gullies should have check dams at the gully mouth where the gully slope merges with that of the

meadow. Discontinuous gullies should also have check dams immediately below the gully heads. Placed immediately below the juncture of two or more gullies, one dam can provide two or more catch basins, thereby increasing its effectiveness. Narrow points of a gully allow a dam of certain height to be built with less material than where the gully is wide. Other key locations for check dams are points of least rapid erosion in the gully owing to a gentle gradient, better foundation material—rock, for example—in the gully bottom, or a protective cover of vegetation.

Check dams should be built as cheaply as possible to realize the greatest investment return. They must be well built to avoid causing additional damage to the meadow. The structure must provide safe passage of flood flows over the dam. This necessitates a low center that draws the overflow toward the middle of the channel, preventing the water from cutting around the dam. In high mountain meadows without large watersheds, making the sides 1 to 3 ft (0.3 to 0.9 m) higher than the center is usually sufficient. An apron below the downstream face of the dam is essential to prevent falling water from undercutting the dam. Check dams are not intended to be impervious. Material fine enough and stable enough to prevent large cracks or pipes opening through the dam should be placed against the upstream face. Such cracks or pipes may pass sufficient water to undercut the dam. The control value of vegetation should be applied to the fullest possible extent. But the catch basin above a check dam should not be filled with loose brush; it tends to remove litter and silt which would otherwise aid in sealing the dam.

Newer check dam designs are now available (Gray and Leiser 1982; Heede 1965, 1966, 1977). Also, computer programs are available (Heede and Mufich 1973, 1974) that specify check dam design, materials, and costs with a minimum of survey work. A prime reason for failure of gully control structures is inadequate attention to design. These new designs and computer programs hold promise of minimizing check dam failure.

Planting Vegetation—A quick and effective means of securing a vegetative cover for the control of soil erosion is the planting of willow cuttings. Willow stakes that hold brush in place may take root, grow, and hold the soil long after the brush has decayed (Gustafson 1937). Undercutting of banks and widening of gully bottoms may occur, however, if channels become choked with willows (Heede 1960).

A significant physiological characteristic of willows is the ability to produce abundant roots from cuttings (Massey and Ball 1944). Freshly cut willow stakes generally take root and grow when set in rich moist soil, as on stream sides and in wet meadows. Twigs broken off are also often able to become established and enlarge the stand. Several willow species grow from cuttings in new road fills and in bare, denuded gullies. Even on unfavorable sites, willow cuttings often grow vigorously for a few years before dying out.

In California, willows are used almost exclusively for cuttings. An ample supply of willow cuttings is usually available in the vicinity of the meadow to be controlled. This is significant in terms of cost as well as the assurance that the species is suited to the locality.

Wherever possible, vigorous native willow species should be used. Species with long, straight stems are easier to cut and drive into the ground than those with crooked stems.

Stakes should be cut and planted when the willows are dormant. This period extends from fall, when the leaves start to turn yellow, to spring, when growth starts. In moist soils, willow stakes are sometimes successfully planted during summer, but this is not recommended.

Little is known of the ability of other woody plants to grow from cuttings without care in mountain meadows. Also, availability may limit the use of other species, even if grown readily from cuttings. Some possible alternatives to willow are mountain alder *(Alnus tenuifolia),* quaking aspen *(Populus tremuloides),* western azalea *(Rhododendron occidentalis),* and huckleberries. These species may be better to plant in some situations, even if it means starting from seed. Huckleberries, for example, inhabit acid soils, but willows cannot tolerate strongly acid soils (U.S. Dep. Agric., Forest Serv. 1937). Where the meadow soil is strongly acid, therefore, huckleberries may grow better than willows.

In general, species that have browse value are preferred. But a plantation of palatable species may be quickly destroyed by livestock. Unpalatable and palatable species should be mixed where the meadow is subjected to livestock grazing or high deer or elk use. Observations should identify the palatable and unpalatable species growing in a locality.

The heavy sod found in mountain meadows is an excellent protection against erosion. Strips of such sod planted in key spots or in contour strips soon spreads and forms a strong cover for the soil. Care should be exercised to dig sod from level places in the meadow where no danger of new erosion exists.

Sods should be planted immediately before or during wet seasons. Where the soil is always damp, sod may be planted anytime. As a rule, obtain sod from the locality where it is to be planted. Plant the sod as soon as possible after cutting. If it is necessary to delay planting for one or two days, keep the sod moist. Sod should be placed with its surface slightly below ground level.

The effects of cuttings and sod plantings can be supplemented by sowing grass or cereal grain. Such stands may last one year only, or become replaced with native species that recapture the site.

EVALUATING RANGE CONDITIONS

Evaluating or classifying range condition is difficult. The methodology is still evolving. A range condition class is "one of a series of arbitrary categories used to classify range condition and usually expressed as either excellent, good, fair, or poor" (Range Term Glossary Committee 1974, p. 21). A fifth

class—very poor—has frequently been used. I suggest that a four-class system is adequate for evaluating most meadows in the Sierra Nevada.

Evaluating range condition—current productivity relative to natural capability—includes a subjective evaluation. Evaluations of range condition will therefore vary. To some, condition is excellent only if herbage production and species composition are near climax. To others, condition is excellent only if calf or steer weight gains produced are maximum.

Equally difficult is detecting condition trend and determining trend direction. Range condition trend is defined as "the direction of change in range condition" (Range Term Glossary Committee 1974, p. 21). Trend is long-term progressive or regressive change. Without change there is no trend. Four main reasons explain the difficulty of determining trend in condition (Reppert and Francis 1973). Trend is evaluated over a period of many years. Different people frequently measure condition trend plots from one measurement to the next. Cause of trends in condition are unnoticed or not documented. And frequently, condition trend is interpreted by persons other than skilled range examiners.

Primary to evaluating range condition and condition trend are the characteristics needed to sustain production of a desired product mix. Range condition methodology aims to detect departure from those characteristics. Condition criteria aim to properly and accurately rate the degree of departure. Trend methodology aims to detect differences between times of observations in the degrees of departures, accurately indicate trend direction, and assess the cause of trend.

The characteristics basic to high range condition, regardless of the desired product mix, are geological and biological stability (Benedict 1981, 1982).

Geologic stability implies a static situation as related to trend in soil condition. Yet, as evident in meadow soil profiles, the stability is dynamic. Geologic instability implies a downward trend in soil condition. A downward trend in soil condition is possible with geologic stability but, unless halted may lead to geologic instability. If a meadow is not geologically stable, biologic stability is unattainable and management can do little to halt the downward trend in condition.

Biologic stability may occur at a stage below the climax potential and therefore does not necessarily connote excellent or good condition. Biologic instability implies condition trend toward or away from the potential. Directing condition trend to attain biologic stability with those characteristics required to meet management goals or objectives is the task of the land manager. For the most part, management goals or objectives are served well if biologic stability is attained and maintained in an open meadow environment with herbaceous vegetation composed mostly of climax perennials.

Established Methods

Certain methods field tested over many years can guide managers evaluating meadow condition and condition trend. The three-step method and the species composition method are two of these.

Three-Step Method

The three-step method of range condition and trend analysis (Parker 1954) has been widely applied on Forest Service grazing allotments throughout the Western United States. It has also been used by the Bureau of Land Management and the National Park Service, U.S. Department of Interior.

The studies that led to the development of the three-step method of assessing condition and trend were carried out by the Forest Service's Division of Range Research, Division of Range Management, six western Regions, and the Forest and Range Experiment Stations. The studies were started in 1948. In 1949, the three-step method was first tested; subsequently, the method was revised and retested in 1950. In both years, the method was tried on various types of ranges and in varying degrees of condition.

The persons first testing the three-step method—Kenneth W. Parker and his colleagues (Parker 1954)—were skilled range examiners. And the three-step method was intended for persons of similar skill.

The three-step method consists of the following:

Step one—Establish permanent line transects and record and summarize the data obtained from them. Establish transects only on the primary range and on sites representative of major condition classes; do not establish transects at random. Lay out each transect so that it is 100 ft (30.5 m) long in the center of a 150-ft by 100-ft (45.7-m by 30.5-m) plot.

Place transects in clusters to obtain a larger sample, a measure of variation, and more useful information per manhour. A cluster should contain a minimum of three transects when the plant density index is less than 30, two transects when the index is 30 to 60, and one transect when the index is more than 60. Use a minimum of two clusters for each major condition.

When the majority of plant species are easily identified—usually in the growing season—make observations at 1-ft (30-cm) intervals along a transect. Use a 3/4-inch (1.9-cm) diameter loop, and classify the area that the loop delimits as erosion pavement, bare soil, vegetation, litter, or rock. Record observations on a specially designed form. Classify plant species as desirable or primary, intermediate or secondary, and undesirable or low value. The form is designed to separate low-value species from the more desirable ones.

Plant density index is the total of all hits within the 3/4-inch loop with established perennial vegetation. Forage density index is the total hits with either primary or secondary species. Ground cover index is the number of hits on bare soil subtracted from 100. Record the number of hits on each plant species.

Measure plant vigor randomly by recording the leaf length on 10 plants of valuable species within the plot.

Step two—Summarize and analyze the data for the cluster, classify the current condition, and estimate current trend. Record data on a "cluster summary" form—primary, secondary, low value.

Record the average number of hits, and the average percentage of total plant density (species composition) for the key species of the three vegetative classes.

Determine current vegetative condition from the sum of the forage density index, composition, and vigor ratings. Determine the condition from a score card prepared for the specific range type.

Rate erosion hazard and current erosion factors. Sum the ratings, then match the sum against ratings for soil condition classes to assess soil stability. Determine erosion hazard from the ground cover index that is rated from 0 to 15. The rating increases with the number of nonsoil hits. Determine current erosion from five defined erosion classes that rate erosion from 0 to 15, depending upon severity.

Determine current trend in forage condition from trend standards (prepared for each condition class) for the particular range type. The standards are based upon plant vigor and forage utilization. Vigor is given twice the weight of utilization. Record trend as up, down, or static.

Determine the current trend in soil stability from standards for condition classes. The variables used for determining trend in soil stability include the amount of litter being replaced each year, visible erosion, trampling displacement, rodent activity, and healing in gullies. As with trend in forage condition, record trend as up, down, or static. The information obtained in steps one and two can be used in subsequent years to determine the long-term trends in vegetation and soil conditions.

Step three—Take two key photographs. Take one general type from one end of the transect. Compare this photo with photos taken from the same position in past or future years to show general changes. Take an oblique close-up photo of a 3-ft² (0.9-m²) plot. Take this photo and subsequent oblique close-up photos from the same point as the general one.

Species Composition Method

The Soil Conservation Service, U.S. Department of Agriculture, has developed and used the species composition or climax method (Bell 1973, Dyksterhuis 1949) of range condition analysis most extensively. Condition trend is not directly determined as in the three-step method. Compositional changes over time, however, are indicators of trend and trend direction.

Primary to the species composition method is recognition of range sites (Range Term Glossary Committee 1974). A range site is a kind of land or a class. In terms of the classification given in this paper, a range site is similar to a meadow site association. The difference is the definition of range sites by potential vegetation and definition of meadow associations by current vegetation. As potential vegetations of meadow sites are defined, the difference is eliminated.

Conceptually, each range site has the potential to produce a unique combination of species and amounts of them, provided physical characteristics have not deteriorated. Condition of a range site individual (Ratliff and Pieper 1982) can be determined if the potentials of the class to which it belongs are known.

Species are categorized as decreasers, increasers, or invaders. As condition deteriorates, the decreasers decrease and the increasers increase in percentage of the composition. With further deterioration, the increasers decrease and the invaders increase significantly. Range site individuals with the composition consisting of 75 to 100 percent climax decreaser and increaser species are considered in excellent condition. The percentage of increasers allowed is limited. The limit depends upon the percentage of a species expected in the climax stand. Any excess of increasers counts against condition and is included as part of the invader percentage. When decreasers and allowable increasers together comprise between 50 and 75 percent of the composition, condition is considered good. Fair condition sites contain 25 to 50 percent decreasers and increasers, poor condition sites contain less than 25 percent decreasers and increasers. The idea of the species categories is defined in the three-step method as primary, secondary, and low-value species.

Stability of the soil is implicit in the species composition method. A specific range site individual must have the potential of the range site to which it is assigned. It loses that potential if the soil is gone.

Condition Standards

Standards of meadow condition should at least be specific to meadow series, and preferably specific for meadow site associations. Attaining either appears distant, and managers need standards. Based on data available on standards and other research results, I suggest that the following conservative, generalized standards are applicable to meadow sites of the Sierra Nevada.

Soil Condition

Given that a meadow is not past the geomorphic slope threshold, and that maintaining or improving its condition is a reasonable expectation, geological stability on specific sites is related to soil stability. Condition is satisfactory if the site potential, as judged by soil characteristics, is maintained (Smith 1979). And soil condition has been given more weight than vegetative condition in assessing condition.

Four condition classes have been described (Ellison and others 1951, pp. 23, 24):

Condition:	Description
Satisfactory condition	"Soil stable under a normal or near-normal amount of vegetal cover; a high proportion of desirable forage plants."
Unsatisfactory condition (a)	"Soil stable under a normal or subnormal amount of vegetal cover; a low proportion of desirable forage plants."
Unsatisfactory condition (b)	"Soil unstable under a subnormal amount of vegetal cover; a high proportion of desirable forage plants."
Very unsatisfactory condition	"Soil unstable under a subnormal amount of vegetal cover; a low proportion of desirable forage plants."

Vegetal cover includes litter (Ellison and others 1951). The third condition can occur with a good stand of desirable

species but with little of the yearly production left. Presence of a good or excellent vegetative condition with a poor soil condition is therefore possible. That situation was frequently encountered on meadows of Sequoia and Kings Canyon National Parks (Bennett 1965). At least in those situations, his evaluations of meadow conditions more strongly reflected soil condition than vegetative condition.

For meadows in satisfactory or good condition, foliar and litter cover should combine so that no bare soil can be seen (Ellison and others 1951).

In Oregon, foliar cover on good condition meadows was 68 percent; on fair, 47 percent; on poor, 31 percent; and on very poor, 12 percent (Reid and Pickford 1946). Amounts of litter and bare soil were not included in the data. However, one excellent condition meadow was reported to have 17 percent bare ground, the suggestion being that surface not covered by foliage was bare. I suspect, nevertheless, that most of the remaining surface, at least on good and fair meadows, had abundant litter.

Excellent condition meadows on the eastern slope of the Sierra Nevada had minimum foliar cover of 70 percent; good condition meadows, 50 percent; fair condition meadows, 40 percent; and poor condition meadows, 25 percent (Crane 1950). Very poor condition meadows had less than 25 percent foliar cover. Litter covered at least 22 percent of the surface of excellent condition meadows, 30 percent of good condition meadows, and 27 percent of fair condition meadows.

Nonplant area was defined as hits on "erosion surfaces, rocks, manure, erosion pavement, bodies of water, and organic litter" (Bennett 1965, p. 24); "an area with no visible plant life within a one centimeter radius of the transect point" (Strand 1979a, p. 79). Strand remeasured several transects originally established by Bennett on meadows in Sequoia and Kings Canyon National Parks. Nonplant areas found by Bennett ranged from 6.4 to 52.6 percent *(table 13).* Foliar covers therefore ranged from 47.4 to 93.6 percent. By the foilar cover criteria of Reid and Pickford (1946) or Crane (1950), all 11 meadow areas were in fair or better condition.

Table 13—*Condition, foliar cover, and nonplant cover at 11 meadow sites in Sequoia and Kings Canyon National Parks, California*

Meadow site	Condition	Cover: Foliar	Cover: Nonplant	Cover: Erosion[1]
		Percent		
Zumwalt	Very good	87.8	12.2	1.2
Upper Paradise	Fair	82.9	17.1	8.7
Arrowhead Lake	Good	79.2	20.8	9.4
Cotter	Good	55.5	44.5	12.5
Fjord Stringer	Fair to good	47.4	52.6	5.3
Charlotte Lake	Very poor	75.5	24.5	8.9
Vidette	Good	93.0	7.0	2.8
East Lake	Fair	81.8	18.2	11.2
Junction (Bubbs)	Fairly good	87.8	12.2	5.6
Junction (Kern)	Very poor	72.5	27.5	4.8
Upper Funston	Poor	93.6	6.4	1.0

Source: Bennett (1965)

[1]Percent of erosion surface included in nonplant cover.

Among the variables comprising nonplant area (Bennett 1965), only the amount of erosion surface was reported. Amounts of erosion surface ranged from 1.0 to 12.5 percent of the surface areas. Assuming that the remaining amounts of nonplant areas were mostly litter, it becomes evident that both Bennett (1965) and Crane (1950) require nearly 100 percent cover as defined by Ellison and others (1951) for a condition classification of excellent.

That same basic standard is adhered to by the U.S. Forest Service, Pacific Southwest Region (U.S. Dep. Agric., Forest Serv. 1969). A maximum of 5 percent bare soil and erosion pavement combined is permitted for excellent soil condition. Amounts may not exceed 25 percent for good condition, 45 percent for fair condition, and 79 percent for poor condition. On that basis, Zumwalt, Vidette, Junction (Kern), and Upper Funston meadows (Bennett 1965) *(table 13)* would be considered excellent in condition. All others would be classified as good.

Other indicators were included in assessing conditions on meadow areas (Bennett 1965): percentage of area trampled, invasion by lodgepole pine, and herbaceous vegetative condition. In effect, these indicators serve the same function as the classification of site damage to determine the soil condition score (U.S. Dep. Agric., Forest Serv. 1969). Damage resulting from all previous causes is classed as severe, moderate, or light. That classification, with the amount of bare soil and erosion pavement, determines the soil condition class.

The standards of Reid and Pickford (1946) and those of Crane (1950) have a foliar cover base. Cover is expressed as the percentage of surface area hidden from view by foliage. The surface area not covered by foliage is proportioned among soil, litter, and other surface variables. The Forest Service's Pacific Southwest Region uses a basal cover base (U.S. Dep. Agric., Forest Serv. 1969). Hits on herbaceous vegetation are recorded at the soil surface. The three-step loop procedure is used, and only one species or surface variable is recorded per loop. Basal cover is expressed as frequency of point (plot without size or shape—a sharp point) hits (Ratliff 1979). Hits on plants are recorded only when actual contact of the point is made with a plant where the plant emerges from the substrate.

I prefer basal cover to foliar cover as a measure of soil condition. Basal cover is affected by compositional changes. Grazing tends to stimulate short plant and moss growth, but basal composition should be little influenced by the current level of grazing, especially if the treatment has been of long duration. Foliar cover is better related to productivity than is basal cover determined by either loops or points. Grazing, however, alters the relationships between foliage and the surface. Except when indicating herbage utilization or preference, therefore, use of a foliar cover standard should be restricted to ungrazed meadows.

The standards for bare soil (U.S. Dep. Agric., Forest Serv. 1969) used by the Pacific Southwest Region appear adequate for nonxeric meadows. Of course, the three-step loop method must be used. The remaining area should be covered by plants, litter, and moss. Minor amounts of gravel, rocks, and

wood are acceptable. Minimum herbaceous plant cover allowed for excellent condition wet meadows is 68 percent; for good, 51 percent; and for fair, 35 percent. For dry meadows the amounts are 56 percent for excellent, 36 percent for good, and 21 percent for fair.

The wet meadow standard may be reasonable for sites of the hanging, lotic, and sunken-concave hydrologic classes. They may also be reasonable for montane meadows with a xeric hydrology. For raised-convex and normal meadow sites, however, the wet meadow standards appear low. The dry meadow standards seem low for montane xeric meadows. But they are likely too severe as cover standards for the xeric short-hair sedge sites of the subalpine. Much effort is required to confirm or reject these hypotheses. I suggest, therefore, that to assess soil condition, the current standards (U.S. Dep. Agric., Forest Serv. 1969) be used until more specific standards can be developed.

Vegetative Condition

The following observations can serve as general guides to meadow vegetative conditions (Reid and Pickford 1946). Meadows in excellent or good condition appear to have a dense, even stand of vegetation. After grazing, such a meadow should give the impression of having been mowed because of the rather uniform forage value of the plant species present. Fair condition meadows appear dense, but unevenly, covered. Poor condition meadows have a distinct patchy appearance. Good condition meadows are not particularly colorful during flowering—scattered forb blossoms are not conspicuous. Fair condition meadows are colorful during flowering—clumps of brightly colored forb blossoms blend with green. Poor condition meadows are very colorful during flowering—patches of green occur among dense colonies of conspicuous, brightly colored forbs.

Because a meadow full of wildflowers is beautiful, these general guides may seem reversed. The first test of a climax is that the dominants belong to the same major life form; and in grasslands, the climax dominants are all grasses or sedges (Weaver and Clements 1929). Grasses have an advantage over subdominant forbs, such that the composition shifts away from grasses only by severe disturbance (Clements 1920). Better condition meadows should have fewer forbs to produce flowers than poorer condition meadows. Such was the situation on subalpine grasslands in Washington and Oregon (Pickford and Reid 1942). Also, free choice or season-long grazing did not decrease the abundance of meadow wildflowers (Ratliff 1972).

The second condition class of Ellison and others (1951) is encountered in mountain meadows where the soils are stable. With stable soil, an evaluation of condition depends upon the relative biologic departure of the site from a standard. The frequently accepted standard for excellent condition is biologic stability at or near climax. Species composition is the usual indicator for departure from climax. Species evaluations used in the three-step method generally tend, however, to reflect grazing values. Even with the species composition method, the terms frequently have reflected grazing value more than ecological position (Smith 1979). Although some species of high ecological position also have high grazing value, when grazing value is used, that fact must be made clear to avoid confusion about the meaning of the standard.

Table 14—*Percentages of composition for primary, secondary, and low-value species by vegetative condition*[1]*—averages from 11 meadow sites in Sequoia and Kings Canyon National Parks, California*

Species class	Vegetative condition			
	Excellent	Good	Fair	Poor
	Percent			
Primary	61.9	55.1	41.0	53.0
Secondary	25.2	10.7	10.0	0.7
Low-value	12.9	34.2	49.0	46.3

Source: Bennett (1965)

[1]Standards used were those of the U.S. Department of Agriculture, Forest Service (1969).

The three-step method uses the concept of the species composition method. For a rating of excellent condition, when low value species are absent, primary species must make up at least 75 percent of the composition (U.S. Dep. Agric., Forest Serv. 1969). The maximum allowable amount of secondary species in a climax community must therefore be 25 percent. Minimum combined amounts of primary and secondary species allowed are 68.6 percent for excellent condition wet meadows, 54.6 percent for good, and 25 percent for fair. Allowing 25 percent secondary species, minimum acceptable amounts of primary species for the condition classes, therefore, are 43.6 percent for excellent, 29.6 percent for fair, and 0.0 percent for poor. The species composition method accepts 50 percent primary species for excellent, 25 percent for fair, and 0 percent for poor.

I evaluated the vegetation and cover data from the 11 meadow sites studied by Bennett (1965) and rated their vegetative conditions. The standards for wet meadows (U.S. Dep. Agric., Forest Serv. 1969) were used. My evaluations did not agree fully with those of Bennett (1965). The results provided a useful comparison.

The average percentage composition for primary species *(table 14)* decreased from excellent to fair condition. And the percentage of low-value species increased. Bennett (1965) clearly stated his use of species' grazing values, and many of his low-value species are increasers rather than true invaders. Were the ecological classification used, it is probable that the secondary species would show a marked increase with declining condition—at least through fair condition.

The meadow sites that were classed as excellent had 87 percent primary and secondary species, good had 66 percent, and fair had 51 percent. I suggest, therefore, that excellent condition meadows will indeed have high percentages of primary or decreaser species. The maximum of 25 percent secondary or increaser species appears adequate, at least with the three-step method.

Percentages of decreaser, increaser, and invader species *(table 10)* were determined for the 90 meadow sites referred to

Table 15—*Vegetative condition class*[1] *and vegetative series for 90 meadow sites of the Sierra Nevada, California*

Vegetative series	Sites, by vegetative condition			
	Excellent	Good	Fair	Poor
Agrostis (Bentgrass)	—	2	1	—
Calamagrostis breweri (Shorthair)	7	1	—	—
Carex exserta (Short-hair sedge)	4	—	—	—
Carex nebraskensis (Nebraska sedge)	3	2	—	—
Carex rostrata (Beaked sedge)	5	—	—	—
Deschampsia caespitosa (Tufted hairgrass)	2	3	3	—
Gentiana newberryi (Newberry gentian)	—	2	1	—
Heleocharis acicularis (Slender spikerush)	1	—	1	—
Heleocharis pauciflora (Fewflowered spikerush)	—	3	10	—
Hypericum anagalloides (Tinkers penny)	2	8	9	—
Muhlenbergia filiformis (Pullup muhly)	—	1	7	—
Poa (Bluegrass)	—	1	2	—
Trifolium longipes (Longstalk clover)	—	—	4	—
Trifolium monanthum (Carpet clover)	—		5	—
Total	24	23	43	00

[1]Assigned on the basis of species composition method.

by Ratliff (1982). Vegetative conditions of the sites were evaluated using the standards for the species composition method. A maximum of 25 percent increasers was allowed.

Excellent conditions *(table 15)* were found for 24 of the sites, good for 23, and fair for 43. Two vegetative series, beaked sedge and short-hair sedge, occurred on nine sites, all in excellent vegetative condition. Two common characteristics are responsible for that result. The series represent the environmental extremes. Beaked sedge sites are largely lotic and short-hair sedge sites are largely xeric in hydrology. Both tend to be monospecific, with beaked sedge and short-hair sedge making up at least 50 percent (and usually a much higher proportion) of the composition. Sites of other series tend to have many species. On the sites in excellent condition, the combined amounts of two or three primary species make up more than 50 percent of the composition.

The species composition method was inefficient in separating fair from poor condition sites. None of the sites rated poor in vegetative condition *(table 15)*, owing to the relative lack of invader species. The greatest amount of invaders on a site was 24.3 percent. The average amount of invaders on the 90 sites was 2.1 percent. Where the proportion of the composition made up by decreaser species was low, the proportion of increaser species was at least 25 percent, thereby giving a rating of fair condition. Also, when the percentage of decreaser species was at or just above 25 percent, the increaser species percentage was usually enough to give a good condition rating.

The percentage of increasers allowed is based on the concept that an amount equal to the maximum expected in the climax is normal. Regardless of the amount of decreaser species present, therefore, an amount up to the percentage of increasers normally expected is added to the decreaser percentage in determining condition. I propose that where invader species are few, as in Sierra Nevadan meadows, the percentage of increaser species allowed should be reduced for good and fair conditions *(table 16)*. This has the effect of raising the minimum amount of decreaser species acceptable while keeping the usual condition standards. The proposed standards were used to rate the 90 meadow sites (Ratliff 1982), with these results: 24 rated excellent, 20 rated good, 24 rated fair, and 22 rated poor in vegetative condition.

For rating vegetative condition, I have used nearest shoot-to-point (or closest individual) species composition. But I found the technique biased and suggest actual basal hits instead. Basal hits are affected less by current grazing than species composition or cover based on foliar hits. Foliar composition is, however, the better measure on ungrazed meadows.

Trends

Current condition tells only how a meadow site measures up to a set of standards. Assessment of condition trend tells how well management measures up.

Table 16—*Proposed generalized vegetative condition standards for meadow sites of the Sierra Nevada, California*

Vegetative condition	Minimum decreasers[1]	Maximum increasers[2]	Decreasers and increasers[3]
Excellent	50	25	75 to 100
Good	30	20	50 to 75
Fair	10	15	25 to 50
Poor	—	—	0 to 25

[1]Minimum percentage of composition allowed.

[2]Maximum percentage of composition allowed; excess percentage contributes to amount of invader species.

[3]Range in percentage of the composition of decreaser and allowed increaser species permitted for the condition class.

Condition trend is frequently rated as up, down, or static. An upward trend in meadow condition or static trend with an excellent condition is the usual goal. But detection and prevention of a downward trend in meadow condition is usually emphasized.

A downward trend in condition is first indicated by overuse in association with seven additional variables (Bell 1973):

1. Weakened condition and lowered vigor of decreaser species;
2. Decrease in the size and abundance of decreaser species;
3. Appearance of invader species;
4. Subdominance of climax plants in the stand;
5. Reduction in range productivity and livestock production;
6. Appearance and dominance of woody plant species;
7. Deterioration and erosion of soil.

Indicators of an upward trend in conditions are the reverse.

A meadow site may be considered static in trend or biologically stable when significant change no longer occurs under a specific management regime. The regime of management is significant. Altering management—the grazing system, for example—likely requires a series of biological adjustments to balance the site with its altered environment. Until the adjustments are made, change will occur—there will be trend in condition.

Measurement of change is the key to detection of trend. A score of 30 is the maximum potential for either soil or vegetative condition as determined by the Forest Service's Pacific Southwest Region (U.S. Dep. Agric. Forest Serv. 1969, sec. 480). "The magnitude of change required to indicate a real trend, upward or downward, must be one-quarter or more of the difference between the original (or previous) measurement and the maximum potential." That standard was derived from the fact that the standard deviation of a normal distribution is approximately equal to the range in values divided by 4 (Dixon and Massey 1957). A site with a score of 1 (very poor condition) at the original measurement, therefore, would need a score of $[(30 - 1)/4] + 1 = 8.25$ at a later measurement for trend to be present. A site in good condition with a score of 26 would need a change in its score of 1 for trend to be present. As condition improves, trend is indicated by smaller changes in scores. That feature reflects the need to quickly detect downward trend and correct management.

It may be possible to use that standard to decide when trend is indicated by the species composition method. If the sum of decreaser and increaser percentages were 40 percent on the first reading, at the next reading their percentage would need to be $[(100 - 40)/4] + 40 = 55$ for an upward trend to be declared. The statistical correctness of the standard needs to be investigated, however.

A five-phase procedure for determining trend from data obtained by the three-step method has been suggested (Reppert and Francis 1973). Whether the three-step or some other method is used, I recommend careful study and use of the procedure as follows:

- Correctly execute the three-step or other method at each observation.
- Determine current condition and tentatively assess trend in the field.
- Statistically compare plant group and species data and relate the results to changes visible in photographs.
- Compare statistical and photographic evidence with the trend assessments made in the field.
- Consider all available information and assign a cause for the trend.

MANAGEMENT CONCEPTS

My basic concept of meadow management is that good meadow management demands good range management. There are six requirements that must be satisfied.

(1) Good range management for meadows requires trust, agreement, and commitment. Mutual trust among managers and users, agreement between them on actions to be taken, and commitment of all involved to accomplishing the actions are essential to success of any management plan. Without trust, there will be no agreement. Without agreement, there will be no commitment. And without commitment, nothing will be done.

(2) Good range management for meadows requires establishing reasonable, attainable objectives or goals. Prerequisite is deciding what the basic goal of meadow management should be. Based on the cooperation of persons concerned and knowledgeable about meadows and meadow problems, a basic goal has emerged. Mountain meadows of the Sierra Nevada should be managed in a manner that maintains or restores their ecological integrity while providing products for mankind. Ecological integrity means biologic and geologic stability (Benedict 1981, Benedict and Major 1982) rather than climax equilibrium.

(3) Good range management for meadows requires proper use (Range Term Glossary Committee 1974). Proper use of meadows may be defined as a degree and time (period and frequency) of use of meadow and watershed resources which, if continued, either maintains or restores meadow ecological integrity and is consistent with conservation of other natural resources. Use here is not restricted to livestock grazing but includes all herbivority and all direct activities of mankind.

To attain proper use, managers and users must decide upon the product mix wanted from and the degrees of use acceptable on a given management unit (allotment) or natural land unit (watershed). They must also decide the period(s) during which the product(s) are produced and the frequency of production.

The degree, period, and frequency of use are extrinsic factors controllable by man. When they are correctly coordinated, use of meadow resources will not exceed threshold levels that could cause damage. And the potential for damage from other extrinsic factors is lowered.

The product mix produced may include livestock, timber, big game, small game, songbirds, rest and relaxation, and flowers—anything people want from the unit. A definable carrying capacity exists for each product desired. The carrying capacity for livestock depends upon the resources available to produce that product. When needs for resources overlap, as for cattle and deer, they must be allocated among the products that require them. The optimum carrying capacity for a management unit "expresses the greatest return of combined products without damage to the physical resources" (Heady 1975, p. 115).

Increasing production of one product without decreasing production of other products dependent upon the same resources results in overstocking. Overstocking, in turn, results in overuse of resources needed for production of dependent products. Continued overuse (overgrazing) induces change by altering threshold levels of other extrinsic factors and threshold levels of intrinsic factors, thereby resulting in meadow deterioration.

Nature sets the periods and frequencies of use for wildlife. Man, within limits imposed by nature, sets the periods and frequencies of use for himself and his domestic animals. Improper periods of use by livestock are often manifested as trampling damage which breaks the meadow sod. Calendar dates when conditions are suitable for individual uses vary from year to year. They should be used only as guides.

Unless livestock movements are restricted, their developed patterns of use will continue and be similar from year to year. Where possible, specialized grazing systems (such as two- and three-unit deferred rotations) should be employed. But many grazing allotments on National Forests in the Sierra Nevada have rough terrain and elevations that vary greatly. Those conditions make fencing for, and management of, specialized grazing systems very difficult. Under such situations, attention to animal distribution and stocking should be stressed.

(4) Good range management for meadows requires restoration efforts, where they are likely to succeed. Without geologic stability, restoration efforts may fail to give expected recovery. Therefore, geologic stability should be assessed during planning. Possible effects of present management should also be assessed. Degrees of use, periods of use, and frequencies of use that are not compatible with planned actions make success of restoration efforts improbable.

(5) Good range management for meadows requires determination of condition and monitoring of condition trend. Meadow productivity declines with decreasing condition, and reduction in productivity is proportionally greater between the lower condition classes. Therefore, management and individual user goals or objectives will generally be well served if a stable herbaceous vegetation composed mostly of climax perennials is maintained in an open meadow environment.

(6) Good range management for meadows requires user education. People disturb meadows. Users should learn how to properly use meadows and their associated ecosystems for the product(s) they desire. They should learn to work with other users and managers to attain the basic management goal.

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This state-of-knowledge report summarizes the best available information on maintenance, restoration, and management of meadows of the Sierra Nevada, California. Major topics discussed include how to classify meadows, meadow soils, productivity of meadows, management problems, and how to evaluate range conditions and trends. Current methods and standards are reviewed, and revised standards for evaluating conditions and trends are suggested. A primary conclusion is that proper meadow management requires proper range management, including the watershed above meadows and control of user populations.

Retrieval Terms: meadow classification, meadow productivity, meadow management problems, meadow conditions and trends, mountain meadows, Sierra Nevada, California

United States
Department of
Agriculture

Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-85

Guide to Testing Insecticides on Coniferous Forest Defoliators

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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

December 1985

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CONTENTS

INTRODUCTION

Coniferous forests of the Western United States are periodically infested by epidemic populations of defoliating insects. Effective and safe use of insecticides, an integral part of most pest management programs, is one method of control.

Inherent in the insecticide application process, however, is an extremely high degree of variation. Chemicals used may be ephemeral or persistent; one species of insect may be susceptible to a particular chemical while another may not; there may be differences in susceptibility or tolerance to a specific chemical insecticide among populations of an insect species; the pest's vulnerability and susceptibility may vary at different developmental stages; weather conditions in the test areas may differ, together with forest type and stand structure; dosage rate, type and volume of formulations vary with insecticide; spray nozzle systems and aircraft types are many; accurate deposit analysis depends on an understanding of the physics of droplet spectra and impingement, and these differ with insecticide formulations and test conditions.

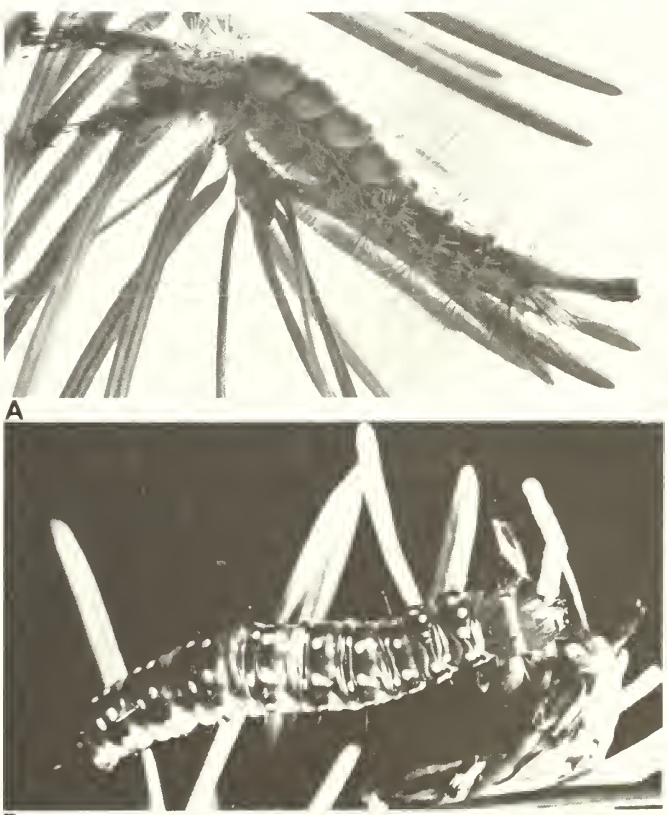
Human variation plays its role as well. Entomologists have varying degrees of experience in planning and supervising research, pilot and control projects, and in analyzing and interpreting the data.

Many sources of variation can be better controlled by careful planning. In developing plans to test insecticides, the manager must rely on the best available information. Some information on biological evaluations about insect outbreaks and control operations has been published, but much of it exists only in unpublished reports and, therefore, is not easily accessible. We have sought to bring together in this report much of the information necessary for planning.

This report provides a guide to testing insecticides applied to coniferous forest defoliators. It outlines techniques for designing, installing, conducting, and evaluating various types of projects, and describes the sampling considerations, methods, and analytical methodology necessary to do the job.

This guide is based both on our experiences in the forests of Washington, Oregon, Idaho, and Montana, with populations of western spruce budworm (*Choristoneura occidentalis* Freeman) and Douglas-fir tussock moth (*Orgyia pseudotsugata* McDunnough), and in the Northeastern United States with populations of the spruce budworm (*Choristoneura fumiferana* [Clements] Freeman). Although the guide is based primarily on studies of budworm populations, much of the information reported can be applied to other defoliating insects such as the Douglas-fir tussock moth, hemlock looper (*Lambdina fiscellaria lugubrosa* Hulst.), the pine butterfly (*Neophasia menapia* Felder and Felder), and the larch casebearer (*Coleophora laricella* Hubner) (*figs. 1, 2*).

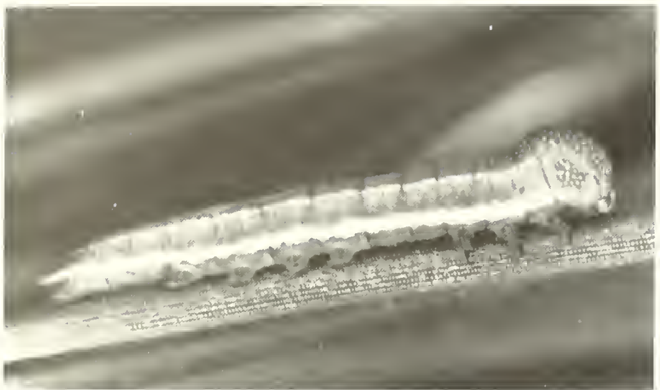
No guide can substitute for work habits and attitudes necessary for a thorough, careful collection of data. Care and thoroughness are particularly vital at the interface between biological phenomena that are observed and data that represent them.



A

B

Figure 1—Douglas-fir tussock moth (*A*) and western spruce budworm (*B*) are the most important defoliating pests of Douglas-fir and true fir forests in the United States. The larvae of both species are similar in their feeding habits and distribution in the host trees. Their populations can be sampled with similar methods.



A

B

Figure 2—The pine butterfly (*A*, larva; *B*, adult) is the most serious defoliating pest of ponderosa pine forests in the United States. The methods described in this guide are useful in projects to suppress epidemic populations of this insect.

EXPERIMENTAL DESIGN

Objectives

The Forest Service, U.S. Department of Agriculture, uses a progression of tests to evaluate and register various insecticide treatments for use in resource protection. These are field experiments, pilot control tests, and operational control projects.

The primary objective of field experiments is to determine minimum effective dosage of one or several treatments against a specific target pest. Treatments may involve different insecticides, different dosage rates or formulations of the same material, or different application equipment. Ensuring adequate replication within the confines of operational constraints, such as manpower and cost, usually requires that treatment plots be small.

Pilot control tests evaluate the most promising treatment identified by field tests under operational conditions. Usually, only one dosage is evaluated and the plots are large enough to simulate operational conditions. Data from field experiments and pilot control tests are used to register highly promising insecticides with the U.S. Environmental Protection Agency. Operational control projects use registered pesticides in a safe manner to achieve resource protection. Resource management objectives are defined in operational control programs and benefit-cost effects of alternative control activities on these objectives are evaluated. These programs may attempt to treat an entire infestation and are carried out on large areas, broken up into spray blocks of thousands of acres. Because their objectives differ, the design and plan for each will differ. Nevertheless, many considerations and activities are similar among them.

In planning a field experiment, a pilot control test, or an operational control project, the first step is to state the objectives clearly, concisely, and as specifically as possible. The objectives may be to estimate treatment effects or specifications to be met, or to test hypotheses. Some field tests of insecticides may be planned to determine the lowest effective dose to achieve a certain percent control or residual population for each treatment and to compare treatments. Usually, experiments with such general purposes are evaluated by analysis of variance (ANOVA). If overall treatment effects are indicated as significant by the F test, tests for differences between pairs of means are the next focus of interest. In other tests and control projects, more specific purposes may be to estimate percent reduction or initial as against residual population size.

To arrive at an effective design for either hypothesis testing or estimation, we must consider variability and cost. Because the criteria for judging effectiveness for the two goals are different, the design parameters—such as number of plots, number of trees, will likely be different. In some instances, both hypothesis testing and estimation are important in a single field test. The test should then be designed using both criteria. The set of design parameters that both meet design criteria for one goal and exceed the criteria for the other goal should be used.

Hypothesis Testing

In choosing the framework for testing insecticides, we need to consider the power of the tests, that is, the probability of detecting treatment effects when they exist. Power is the criterion by which confidence in the experimental design can be measured.

If we designate the hypotheses

H_0 : no differences exist between treatments, or between treatments and control, or

H_1 : one or more treatments differ significantly from the control or from each other,

then the two kinds of error that can be made in hypothesis testing are Type I or Type II, in which:

Type I error is to decide there is a treatment effect when, in fact, there is none; or to reject a true hypothesis (accept H_1/H_0)

Type II error is to decide there is no treatment effect when, in fact, there is one; or accept a false hypothesis (accept H_0/H_1).

Results of possible decisions in hypothesis testing are as follows:

		Correct Decision	
		H_0	H_1
Actual Decision	H_0	No error $P = 1\text{-}\alpha$	Type II $P = \beta$
	H_1	Type I $P = \alpha$	No error $p = 1\text{-}\beta$

The size of the Type I error is referred to as α and the size of the Type II error as β. The power of the test is $1\text{-}\beta$, or the probability of rejecting a false hypothesis. In the field experiment, $1\text{-}\beta$ is equivalent to the probability of detecting a treatment effect when one exists.

Suppose, for example, we are interested in detecting a difference of 10 percent in number of larvae killed between two doses of an insecticide, and that we erroneously decide that there is not this much difference when, in fact, there is a difference of more than 10 percent. Then a Type II error has been made. If the probability that a Type II error will occur is greater than or equal to 20 percent, given the design of our test procedure, the power of this test is less than 80 percent. Keeping the power above a given level ($1\text{-}\beta$) requires keeping the Type II error below β. We always want to maximize the power and minimize the probability of making a Type II error, given an initial choice of acceptable size for the Type I error, α.

Variance, cost, and experimental design influence power. For a given sample size, the larger the variance within or between experimental units, or both, the lower the power. Consider, for example, an experiment in which n trees on a plot are sprayed with an insecticide and n trees on another plot serve as controls. If the number of insects, after spray, varies considerably from tree to tree (which is usually the situation after treatment), it is more difficult to detect if the treatment has an effect. The variance is larger than it would be if all trees within a plot had the same postspray count (low variance).

The relation of cost to power concerns sample size, because increased sample size reduces variance, leading to increased power. A problem then confronting the investigator is choosing the maximum possible sample size for a cost that is to be held at a given level.

Factors in experimental design that influence power include number of levels per chemical, the spread of differences between levels, number of replications, and number of sampling and subsampling units. Power can be increased by decreasing the number of treatments and by selecting treatments for comparison with expected large differences in treatment effects. For a given number of observations, power increases as number of observations per treatment increases.

Costs

The cost of sampling is, of course, a primary constraint on sample size, and must be considered when the experiment is designed. A cost function can be constructed to take into account both fixed and variable costs. Fixed costs, such as rental of aircraft, do not fluctuate particularly with sample size, but costs for plot selection, layout, and sampling depend on worker hours. Cost of personnel will vary with number of units and subunits sampled, and with other factors peculiar to the sampling situation, such as roughness of terrain, weather conditions, and accessibility and condition of roads.

A typical cost function for this type of experiment would include items such as the following:

1. Aircraft cost
2. Experimental design and plot selection
3. Travel time between worker's residence and study areas and between plots
4. Plot layout, flagging and corner marking, and removal of markers
5. Selection, marking and mapping of trees, travel between trees on plot
6. Setting out and picking up spray cards associated with study trees
7. Reading of spray cards in the laboratory
8. Field collection of samples (branches), number of branches per tree, travel between trees on plot
9. Prespray examination of samples in the laboratory
10. Postspray examination of samples in the laboratory.

The cost for item 1 depends on carrying capacity and type of aircraft, distance between landing strip and experimental plots, number of plots and number of acres per plot, distance between plots, and topographic features of plots. The cost of items 2, 3, and 4 depend upon number of worker-hours per plot, and items 5 and 6 upon number of worker-hours per tree. Item 7 may be a fixed cost per spray card (and therefore per tree), while items 8, 9, and 10 depend on number of worker-hours for handling branches. Costs per worker-hour will vary according to the nature of the task, but can usually be entered as an estimated constant. If terrain, accessibility, and other conditions vary much from plot to plot, initial plot selection and insect population estimates may have to be done separately by plot.

Data Analyses

To account for treatment differences, given natural differences and sampling error, we can define the variables of interest to be the ratio of insects to foliage after treatment to that before treatment. An estimate of this variable and the components of the variances of the estimates can be computed on the basis of ratios of the plot means. Usually, the experiment will be designed to test differences between dose levels, or between treatments and control, for a number of insecticides. The model for analysis of the experiment will then be evaluated by a one-way ANOVA with plot ratios as observations and insecticides at various levels with controls as treatments.

Variability in sampling within such an experimental framework is often so high, however, that the power of tests of differences within the bounds of feasible sampling configurations is not high enough. It may have been decided ahead of time, for example, that in testing a hypothesis of no difference between treatment and control no more than 21 trees on each plot with two branches per tree can be sampled

because of cost and other considerations. Yet, the variance estimate for such a sampling configuration may be such that the probability of detecting a desired 10 percent difference between treatment and control would be only 50 percent.

We will first develop and illustrate the design procedure for one-way ANOVA with randomized blocks and then provide an alternative framework which, when appropriate, can somewhat alleviate the need for large sample size.

The process of design consists of selecting a set of design parameters that maximize the power of the F test of the hypothesis H_0: $B_1 = B_2 = \ldots B_r$ against the alternative hypothesis H_1: $B_1 = \beta_i$, $i = 1,\ldots,r$. The B_i are the treatment effects, in our example, the true mortality under treatment i. The parameters we can set are:

r = number of treatments
nb = number of blocks (replications)
nt = number of trees per plot
nbr = number of branches per tree

To maximize power we need to calculate the noncentrality parameter ϕ of the noncentral F distribution:

$$\phi = \frac{\sqrt{\sum_{i=1}^{r} \beta_i^2}}{\sigma\sqrt{r/nb}}$$

One of the variables, nb, in ϕ is a member of our set of design parameters that will be varied in searching for a maximum. Of the other three variables, the σ must be estimated from data on hand, the β_i's set to reflect our alternative hypothesis, and the r is given when we set our objectives.

The usual error mean square from the one-way ANOVA with randomized blocks is used to estimate σ^2:

$$\hat{\sigma}^2 = \sum_{i=1}^{nb} \sum_{j=1}^{r} \frac{(y_{ij} - y_{i} - y_{j} + y_{..})^2}{(r - 1)(nb - 1)}$$

in which

y_{ij} = after/before ratio of number of insects for block i, treatment j (the number of insects is totaled over all branches and trees on each plot)

$y_{i.}$ = mean for block; $y_{.j}$ = mean for treatment; and $y_{..}$ = overall mean.

This shows $\hat{\sigma}^2$ as a function of nb and r, but not explicitly as a function of nt and nbr. To let the number of trees and branches vary in our search for maximum power, therefore, we must break σ^2 into its components; the variability between plots, σ_e^2, and the sampling variance in estimating each plot mean, σ_s^2. Specifically, experimental error $= \sigma^2 = (nt)(nbr)\sigma_e^2 + \sigma_s^2$. The sampling variance in estimating each plot mean is estimated (Kendall and Stuart 1963, p. 247) by:

$$\hat{\sigma}_s^2 = \left(\frac{E(X2)}{E(X1)}\right)^2 \left\{\frac{Var(X2)}{E^2(X2)} + \frac{Var(X1)}{E^2(X1)} - \frac{2Cov(X1,X2)}{E(X1)E(X2)}\right\} = Var\left(\frac{(X2)}{(X1)}\right)$$

in which

X1 = number of insects per plot before treatment
X2 = number of insects per plot after treatment.

We can use sample estimates for each component. The formulas for estimating two-stage sampling variance are from Cochran (1963, p. 277). For both X1 and X2 we have:

$$\hat{Var}(X) = \frac{s_1^2}{nt} + \frac{s_2^2}{(nt)(nbr)}$$

$$s_1^2 = \frac{\sum_{j=1}^{r} s_{1j}^2}{r(nt - 1)}, \quad s_2^2 = \frac{\sum_{j=1}^{r} s_{2j}^2}{r(nt)(nbr - 1)}$$

$$s_{1j}^2 = \frac{\sum_{k=1}^{nt} (x_{k.} - x_{..})^2}{nt - 1}, \quad s_{2j}^2 = \frac{\sum_{k=1}^{nt}\sum_{m=1}^{nbr} (x_{km} - x_{k.})^2}{nt(nbr - 1)}$$

in which

x_{km} = insect count on branch m, tree k, plot j
$x_{k.}$ = mean count for tree k, plot j
$x_{..}$ = mean count on plot j

E(X) = grand mean over all branches, trees, plots

$$= \frac{\sum_{i=1}^{r}\sum_{j=1}^{nb}\sum_{k=1}^{nt}\sum_{m=1}^{nbr} x_{ijkm}}{r(nb)(nt)(nbr)},$$ in which x_{ijkm} is the count on branch m, tree k, block j, treatment i.

An unbiased estimate of

$$COV(X_1,X_2) = \sum_{i=1}^{r}\sum_{j=1}^{nb}\left(\frac{\sum_{k=1}^{nt} x_{1ijk.}x_{2ijk.} - (nt)x_{1ij..}x_{2ij..}}{r(nb)(nt)(nt - 1)}\right)$$

in which $x_{1ijk.}$ and $x_{2ijk.}$ are sample means for the kth tree, block j, treatment i

and

$x_{1ij..}$, $x_{2ij..}$ are plot means for treatment i, block j.

Substituting in σ_s^2, we get $\hat{\sigma}_s^2$ with nt and nbr equal to the number of trees and number of branches in the data set used. Then, with $\hat{\sigma}^2$ = error mean square (EMS) from ANOVA and $\hat{\sigma}_s^2$, we get $\hat{\sigma}_e^2$ from $\hat{\sigma}_e^2 = \dfrac{\hat{\sigma}^2 - \sigma_s^2}{(nt)(nbr)}$.

We now have estimates of everything we need to get an estimate of σ^2 for any combination of nt and nbr. We use s_1^2

and s_2^2 to estimate Var(X1) and Var(X2); these are combined with E(X1), E(X2) and Cov(X1,X2) to obtain an estimate of σ_s^2 we can call $\hat{\sigma}_s^2$ for a particular combination of nt and nbr. Then

$$(nt)(nbr)\hat{\sigma}^2_e + \hat{\sigma}_s^2 = \hat{\sigma}^2,$$

an estimate of between-plot variation for the chosen values of nt and nbr.

We now need only to select values of β's to calculate power for a range of design parameters. Reasonable values of differences between the β_i's under alternate hypotheses in this problem can be taken as between 0 and 1. If, in an experiment with two levels of treatment, we wish to reliably detect differences when the pattern of treatment mortalities is at least as different from equal as $\beta_1 = 0.8$, $\beta_2 = 0.9$, then the β_i are calculated as follows:

$$\beta_2 = \beta_1 + 0.1$$

$$\text{with } \sum_{i=1}^{2} \beta_i = 0$$

$$\beta_1 + (\beta_1 + 0.1) = 0$$

$$\beta_1 = 0.05$$

$$\beta_2 = -0.05$$

We now have everything we need to calculate power. For example, if r = 2, nb = 8, $\hat{\sigma} = 0.10$, $\alpha = .05$, then

$$\phi = \frac{\sqrt{\sum \beta_i^2}}{\sigma\sqrt{r/nb}} = \frac{\sqrt{0.05^2 + 0.05^2}}{0.10\sqrt{2/8}} = 1.41$$

The degrees of freedom are 1 and 7, the same as those for the F test for treatment, namely r − 1 and (nb − 1)(r − 1). From charts by Pearson and Hartley (1951), we find for $df_1 = 1$, $df_2 = 7$, and $\phi = 1.41$, the power is 0.41.

When the cost function has been defined in terms of r, nb, nt, and nbr, power and cost can be computed for enough values of these parameters to construct isopower and isocost curves.

To illustrate, some of the results of designing new field tests for Orthene against larch casebearer are based on variance and cost data from an earlier test (Hard and others 1979).[1] A question often asked is whether ½ lb of Orthene is as effective as 1 lb of Orthene. The null hypothesis is that ½ lb yields the same result as 1 lb, that is, $\beta_1 = \beta_2$. To justify the expense of the field test, we would like the probability of rejecting the null hypothesis if ½ lb gives at least 0.1 less mortality than 1 lb ($\beta_1 - \beta_2 \geq 0.1$). The β_i's for the alternate hypothesis can be calculated from $\beta_1 - \beta_2 = 0.1$, $\beta_1 + \beta_2 = 0$. From this we obtain $\beta_1 = 0.05$ and $\beta_2 = -0.05$. We then calculate power and cost for a number of combinations of nb and nt, with nbr set to 2, 4, 6, and 8.

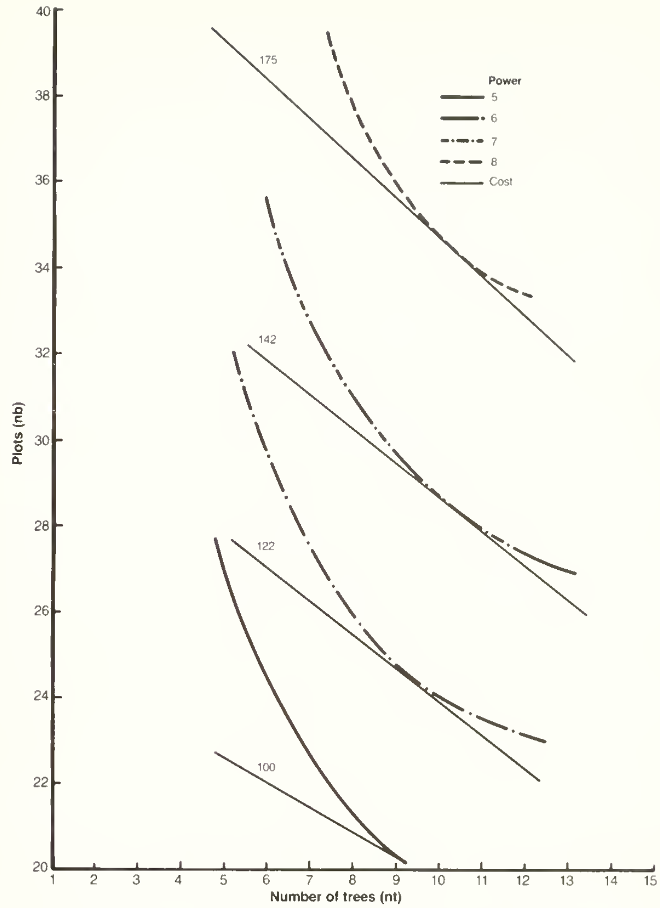


Figure 3—Isopower and isocost curves for various integer values of number of trees per plot or block (nt). The combination of number of blocks or plots (nb) and nt that provide a given level of power for the least cost is found at the point where a cost curve is tangent to the power curve. Cost is expressed as hundreds of dollars.

For a given power, cost is always lowest for two branches per tree (nbr = 2). We then calculate power and cost for all combinations of nb from 20 to 40, and of nt from 1 to 15, with nbr set to 2. Isopower and isocost curves are then fit to power and cost values at the integer values of nb and nt (*fig. 3*). The combination of nb and nt that provides a given level of power for the least cost is found at the point where a cost curve is tangent to the power curve. The point of tangency may not fall on integral values of total number of plots, nt, or both. For power = 0.8, the minimum cost is slightly above $17,500 and could be obtained with 17 replications (34 plots), 11 trees per plot and 2 branches per tree, or 36 plots, 9 trees per plot. For power of at least 0.7, we can choose either nb = 30, nt = 9, or nb = 28, nt = 11. In each situation, the power will be above 0.7, and the cost will be about $14,200. We could, of course, look at the actual cost of each of the combinations and choose the combination with minimum cost. Because we are designing a new study with estimated variances and average costs from previous studies, however, this refinement is not necessary. Either combination available is a reasonable choice if at least 0.7 power is desired. We believe it is unreasonable to plan an experiment to answer an important question with power less than 0.8.

[1] This report neither recommends the pesticide uses reported, nor implies that they have been registered by the appropriate governmental agencies.

As design parameters, therefore, we choose either 34 plots, 11 trees and 2 branches, or 36 plots, 9 trees and 2 branches. But in many instances an experiment of this size is not feasible. Achieving proper time of applications on all plots may be impossible; or, it may be impossible to find enough infested plots in a compact enough area so that the variance used in calculating power still applies; or it may be too expensive.

What can we do in these situations to be reasonably sure of conducting a successful experiment? If we reword our question from "Is ½ lb Orthene as effective as 1 lb Orthene?" to "Are ½ lb and 1 lb each effective?" we may then use a more powerful test. But we must first define effectiveness in quantitative terms. We may believe arbitrarily, for example, that a treatment is effective if the mortality is at least 95 percent, or a treatment is effective if fewer than 5 budworms per 100 buds remain after treatment (McGowan and others 1967, Stipe and others 1977). We may now use a one-sample Student's t-test:

$$t_{n-1} = \frac{(\bar{r} - \mu r)\sqrt{n}}{s_r}$$

in which

r_i = ratio of before and after counts on the i^{th} plot

$$\bar{r} = \sum_{i=1}^{n} r_i$$

$$s_r^2 = \sum_{i=1}^{n} \frac{(r_i - \bar{r})^2}{n - 1}$$

μr = criteria for effectiveness

To design an experiment we again look at the power. With the same number of trees, variance, and alternate hypothesis as above, we calculate the noncentrality parameter of the t distribution as:

$$\delta = \frac{(\mu r - \mu a)\sqrt{n}}{\sigma} = \frac{(0.9 - 0.8)\sqrt{n}}{0.10}$$

in which μa = the mortality under the alternate hypothesis.

With the power curves (Owen 1962) for three values of n we get the following:

n:	δ	Power
6	2.5	0.72
7	2.7	0.78
8	2.9	0.85

To achieve 0.8 power for each of these individual tests of effectiveness, therefore, we need only 8 plots per treatment instead of 17 needed using an analysis of variance design. The variable cost of the experiment is only $8,341, compared with $17,726 for the analysis-of-variance design.

Other Methods

Previous experimental designs for pilot control tests and operational control projects have stressed the reduction of variance in the allocation of resources to obtain good estimates of population densities and mortalities (Mounts 1976). Carolin and Coulter (1972) described a procedure that allocates sampling effort optimally among plots, subplots (clusters), and trees with a multistage computer program of Hazard and Stewart (1974). Analyses are based on variance in numbers of larvae on four twig lots and relative costs attributed to plots, subplots (clusters), and trees.

Some statistical and practical considerations in sampling favor fewer trees per cluster and more clusters per area. A cluster represents a single sample point no matter how many trees are in the cluster. The more trees and branches sampled, the more information obtained about the insect population at that sample point or cluster, which usually covers less than 1 acre (0.405 ha). Beyond a certain point, which depends upon the variability of the data, additional trees in a cluster do not contribute substantially to precision of population estimates. Cluster means are usually more variable than measurements between trees within clusters. In this case, more clusters with fewer trees in each one give a more precise estimate of the population mean. When precision of the estimate is the only criterion for choice of sampling design, single tree clusters are, for the usual case, the most efficient. The advantages of cluster sampling schemes can then be considered in terms of cost and practicality.

Cost per sample unit is the major cost of sampling, and traveling between trees and plots is the major expense incurred. Costs of sampling each tree are essentially similar. Because distances between trees in a plot or cluster are small it may be more economical to choose more than one tree per plot or cluster and thereby reduce the number of plots or clusters in an area. Instead of 60 one tree plots on which to observe treatment effects in a spray area, for example, it may be more cost-efficient to observe treatment effects with 20 three tree plots (clusters). Also, it is easier to locate 20 three tree plots (clusters) on the more accessible sites in a study area than it is to locate 60 plots. The cost-effectiveness of various combinations of clusters and trees is a major part of a computer program by Hazard and Stewart (1974) that allocates sampling effort.

SPRAY AREAS

Boundaries

Our working assumption is that a western spruce budworm outbreak is damaging trees in several National Forests in

Montana. After 3 to 5 years of heavy defoliation, widespread tree mortality will result in one forest if heavy defoliation continues for an additional season. A summer aerial reconnaissance shows large areas of severe defoliation caused by budworm feeding in spring and early summer. A fall budworm egg mass survey indicates continued high larval populations for the next spring. In early fall, research entomologists, with the cooperation of the forest pest management staff and the forest supervisor, decide to conduct field experiments in the infested forests to determine the effectiveness of several promising new insecticides; or alternatively, they decide, in cooperation with the forest supervisor and pest control specialists, to spray those areas where widespread tree mortality is expected and those forests stands to be protected for timber, esthetic high-use areas or winter game range.

A control project (field experiment, or pilot control test) is set up. Responsibilities are delegated. You are the project entomologist. Much of the success of the project depends upon what you do in the next few months.

First, boundaries of the proposed spray area must be determined. Aerial detection and damage reconnaissance surveys are the best sources of infestation damage for determining proposed spray boundaries. Aerial surveys of forest defoliating insects such as the spruce budworm or Douglas-fir tussock moth begin in midsummer after pupation and moth flight are completed. The survey lasts for 3 to 4 weeks and evaluates the current season's damage. Survey procedures, equipment standards and personnel training have been described (Wear and Buckhorn 1955). The procedures allow large areas of visible defoliation to be surveyed quickly and cheaply, and to be accurately mapped. Survey maps broadly delineate proposed spray units in forest areas. Degree of defoliation damage is thought to indicate population levels of the defoliating insect.

Aerial survey maps have limitations, however. They record defoliation damage caused by previous populations but furnish no information for predicting the coming year's defoliations. A biological evaluation to estimate the damage possible from the next generation (that is, the generation that will be treated), therefore, is necessary. This evaluation permits an investigator to determine need and extent of direct control and to draw boundaries around the proposed spray units more precisely.

Reconnaissance

Once boundaries of proposed spray areas are broadly determined, aerial photos and up-to-date maps of the areas are obtained. Aerial reconnaissance is necessary to acquaint the investigator with the topography, forest types, and host and nonhost areas. Infestations occurring in canyons and "hot spots"—most heavily defoliated forest areas—can be identified. Visibly infested areas are outlined on photos with ridge tops, roads, and nonhost areas as boundaries. As much of the infested area as possible should be included within the proposed spray units. Land ownership patterns are determined; state cooperation and clearance from private owners are obtained, if needed. Road systems in proposed spray areas must be known, and quickly identifiable from the air and on the ground. New roads are added to photos and maps after aerial reconnaissance. Natural openings such as meadows, ridgetops, clearcuts, and other areas that are accessible to tank trucks and have potential as heliports are recorded. All roads should be driven, portions of the area that are inaccessible by road should be walked. Infestation, tree conditions, and forest types should be delineated on the maps during reconnaissance.

Spray Blocks

After boundaries of proposed spray areas are determined and the areas reconnoitered, plots are selected for field tests, or the proposed spray area is divided into blocks for pilot tests or operational control programs. A spray block—an operational unit within the proposed spray area—contains similar forest conditions, pest population levels, and developmental stages within the range of variation present in the spray area. Spray block boundaries are selected to coincide with easily recognizable features such as ridges, streams, meadows, or roads.

Spray blocks are usually larger than several thousand acres. In the 1958 Spruce Budworm Control Project in Oregon, spray blocks varied from 106 acres (42.9 ha) to almost 33,000 acres (13,360.3 ha), averaging 7400 acres (2996.0 ha). In the 1974 Cooperative Douglas-fir Tussock Moth Control Project, spray blocks averaged about 1677 acres (679.0 ha). The project consisted of 203 spray blocks organized into seven control units headquartered in nearby towns, from which the various activities on the spray blocks were administered.

Experimental Plots

Plots are usually much smaller than spray blocks, varying in size from approximately 20 acres (8.10 ha) to more than 1000 acres (404.86 ha). To replicate treatments under uniform conditions, plots are located within areas of similar insect population level, developmental stage, and forest habitat type. A spray block may be used as a treatment unit or it may be blocked to contain all of the replicated experimental plots for one study.

Field experiments should provide estimates of the performance of an insecticide that could be expected in an operational control project. Experimental plots, therefore, need to be large enough to include environmental and spray application conditions similar to those in spray blocks of operational control programs. Sufficient area depth is necessary to allow for spray turbulence, drift, and settling within treated areas. Edge effects—which are pronounced in small plots—should be minimized. Small plots require careful spray ap-

plication because application errors can be intensified by greater edge effects. Slight downslope winds (0 to 5 mph [0 to 8 km/h]) in early mornings when spray is normally applied, for example, can cause more than one-third of a 40-acre (16.20 ha) plot to be missed if the pilot does not offset enough uphill to allow for proper downslope air movement.

Field Laboratory

Once spray blocks, experimental plots, or both are chosen, the project director selects the largest town within a reasonable commuting distance to spray areas. The town preferably contains Forest Service facilities and is large enough to provide an adequate local labor supply and provide comfortable living facilities for project personnel.

A building should be located in or near the town for use as a field laboratory. The main functions of the laboratory are to:

- Serve as project headquarters and radio base
- Provide a place for processing field samples and diagnostic work
- Provide suitable conditions for cold storage and laboratory rearing of larvae to determine incidence of parasitism, and laboratory bioassay of specimens collected in the field
- Provide accommodations for examining preliminary data
- Provide for storage of equipment and supplies

SAMPLING

Life Stages

Appropriate life stages—During reconnaissance of the proposed spray area, the insect population is monitored. If the population drops to a low level, particularly before the appearance of the instar to be sprayed, it may be necessary to delete that spray unit in a control program. A field experiment may be canceled because of an inadequate target population. Density of budworm egg masses must be determined in fall and the survival of the overwintering population of small larvae assessed in early spring. Densities of the number of egg masses and hibernating larval populations are directly related to extent of tree defoliation the next spring (Terrell and Fellin 1960).

Evaluating an infestation requires adequate sampling techniques to determine population levels at certain life stages. The various life stages considered to be appropriate sampling periods for the spruce budworm have been thoroughly discussed (Morris 1955). Information can be obtained from examining all periods of the budworm's life cycle: egg masses, hibernating larvae, feeding larvae, pupae, and adult moths. The larval stages of the spruce budworm can be identified from known characteristics of the insect (*table 1*) (Carolin and Stevens 1979, 1981). Population surveys to assess the need for direct control and to evaluate control action have mainly focused on egg masses, hibernating larvae, large larvae, and pupae.

Generally, the small sizes of egg masses and difficulty in finding enough of them when examining samples of foliage make surveys of budworm egg masses time-consuming, expensive, and subject to substantial human errors that must be checked constantly. The egg stage, however, is a period of relative population stability and offers the advantages of a reasonably long period in which to accomplish the work. Also, past and present budworm population levels can be compared directly by distinguishing and counting samples of the current year's and older egg masses obtained from the egg mass surveys of the current year.

A population survey of diapausing (hibernating) larvae also offers the advantage of a long period in which to do the work. This type of survey is difficult, however, and presents special

Table 1—*Head capsule width and general characteristics of spruce budworm larvae, Western United States*

Instar	Average width of head capsule	Characteristics
1	—	Head capsule brown. Prothoracic shield light brown. Body light green.
2	0.35	Head capsule dark brown. Prothoracic shield brown to dark brown and entire body orange brown to yellow orange. Body length above 4 mm.
3	0.49	Head capsule dark brown to black. Prothoracic shield dark brown to black; rear margin slightly undulate in center. Body orange brown with setal areas visible as dots. Body length 5 to 7 mm.
4	0.76	Head capsule black. Prothoracic shield black with slight medial notch on rear margin. Body orange brown; setal areas small and pale; anal shield ivory with brown pattern. Body length 6 to 10 mm.
5	1.26	Head capsule yellow brown with black triangular markings at base. Prothoracic shield black; rear medial notch halfway through shield. Body pale olive brown above, pale yellow brown below with lateral orange stripes. Setal areas a conspicuous ivory color, not raised. Anal shield tan. Body length 10 to 16 mm.
6	2.02	Head capsule chestnut brown to tan. Prothoracic shield divided in the middle and collar-like; same color as head capsule. Body olive brown to dark brown above, orange brown to yellow brown below. Setal areas large, raised, ivory color. Anal shield tan. Body length 16 to 30 mm.

Source: Lyon and others (1972)

problems. Hibernating larvae are surveyed during early spring, a time when forests may not be easily accessible. The work requires some method of selecting sample trees, felling the trees, transporting samples from the forests to the rearing rooms, and placing them in tightly sealed boxes to force larvae to break diapause. All subsequent sampling must be done on a different set of trees with all the attendant problems of intertree variation in the sample estimates.

Most direct control programs of the spruce budworm are aimed at the feeding larvae (3d to 6th instars) life stage. These larvae defoliate trees in spring. Large larvae are highly active and special precautions must be taken to prevent their loss during sampling. The large larval stage (5th and 6th instars) is also a period of high budworm mortality and rapid change in population densities. It is a difficult phase in which to sample large areas in intensive life table studies.

Pupal and prepupal stages of early summer, however, are periods of stability and, therefore, are favorable to sampling of large areas. Pupae can be easily found on foliage samples for rapid and accurate sampling. From live and empty pupa cases data on pupal density, pupal parasites, sex ratio, and moth abundance can be readily obtained. These data are particularly valuable for more completely evaluating direct control treatments than just the data comparing pretreatment and posttreatment population densities of large larvae.

The life stages have been examined to determine their usefulness in predicting subsequent defoliation damage for western budworm infestations in the central and southern Rocky Mountains (McKnight and others 1970), for western budworm in the northern Rocky Mountains (Terrell and Fellin 1960), and for budworm infestations in eastern Oregon and Washington (Carolin and Coulter 1972). In every instance the egg stage was found particularly helpful to making control decisions, and was used for predictive purposes as an index of subsequent defoliation. It is expensive, however, and could be supplemented by analysis of weather from selected weather stations (Hard and others 1980) and a reduced egg mass survey.

In eastern Oregon, old egg masses to represent the previous year's new egg masses can be used to predict budworm population trends (Buffam and Carolin 1966). The infestation trend in the northern Rockies, however, could not be predicted by a single sampling of foliage and recording of old and new egg masses (Terrell 1961).

Sampling universe: trees—In forest insect sampling, it has become common practice to refer to the forest stand or habitat in which the "population of interest" occurs as the "sampling universe" (Morris 1955). It is necessary in any sampling problem, therefore, to define the sampling universe carefully, and thereby avoid the danger of applying conclusions to a broader or more narrow universe than that sampled. The terms sampling universe and target universe have specific meanings in statistics. The *target universe*, or population, is the one to which we would like our inferences to apply: it contains all of the elements of concern, that is, all of the defoliating insects in the stand. The *sampling universe* is that portion of the target universe available for us to sample. The two universes will differ because of practical reasons such as accessibility, ability to count elements, and others. When we speak of the sampling universe in field tests of insecticides, we are usually referring to insects on foliage of open-grown host trees 30 to 50 ft (9 to 15 m) tall that are amenable to sampling with a 35 ft (10.6 m) extendable pole pruner. Tree foliage is the primary habitat for larvae of all forest defoliating insects. It is the habitat for most of the life stages of the spruce budworm and the Douglas-fir tussock moth, except for a portion of the overwintering budworm larvae populations hibernating on tree boles and limbs.

Host tree species of spruce budworm are found in various major forest types, age classes, and stand conditions depending upon the action of past influences on the forest. Substantial evidence indicates that budworm density, survival, and larval development rates vary with tree species, forest types, and stand conditions (Morris 1963, Williams and others 1971). It is a mistake, therefore, to choose sample trees from just one host tree or one forest type, or both, for assessing budworm population trends and status, and also for assessing results of various insecticide and other control treatments. The spruce budworm feeds on five species of conifers (Bean and Waters 1961). Balsam fir (*Abies balsamea* [L]) Mill.) is the preferred host. Its counterpart, the western spruce budworm, has at least 14 acceptable hosts (Carolin and Honing 1972), with these as the preferred hosts: Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco), Engelmann spruce (*Picea engelmanni* [Parry]), grand fir (*Abies grandis* [Dougl.] Lindl.), and subalpine fir (*Abies lasiocarpa* [Hook] Nutt.). In comparison with the spruce budworm on balsam fir, the western spruce budworm on Douglas-fir and grand fir deposit larger egg masses, hibernate farther inside the crown, and show greater diversity in age distribution of feeding larvae (Carolin and Coulter 1972).

Examining the effects of chemical and other kinds of control on budworm or other defoliating insects under a variety of forest conditions requires description of each of the conditions. These may be stand density, species composition, number of canopy layers, aspects, and others. Once the conditions are described, any forest stand characterized by a specific set of conditions will be part of a reasonably homogeneous forest universe for sampling. If the objective of the study requires estimates of budworm population change on an area basis throughout a forest type, however, stratified random sampling within the heterogeneous forest universe is necessary. Whichever the interest is—to examine budworms under a variety of forest conditions or to examine budworms on an area basis—the basic sampling universe is trees. It is necessary, therefore, to know between-tree and within-tree variability of budworm population densities.

More population variability exists between trees, even those growing in homogeneous forest conditions, than exists within a tree. Frequently, intertree variance for a given plot is higher than interplot variances within a homogeneous forest. Also, a correlation exists between successive samples from the same tree (Morris 1955). Taking consecutive samples from the same trees, or remeasuring the same trees when the estimates of

interest trend—change between points in time—has a statistical advantage. Also, precision can be gained through remeasuring the same trees if the correlation between successive samples of insects is high enough. More precise estimates of budworm population densities can be obtained, therefore, by taking egg mass samples and larval and pupal samples from the same trees. An individual tree can withstand only so much twig removal, however, and if a large number of samples are desired, it may be necessary to take them from nearby trees.

Sample units—Although trees are the major source of variation in estimates of budworm population densities, to examine all foliage on a tree as a sample unit is neither practical nor statistically desirable (Morris 1955). Instead, some portion of the foliage within a tree must be considered. In a study to sample spruce budworm populations on balsam fir in New Brunswick, branch surface (the foliated length and width of the whole branch) was more suitable as a sample unit than were twigs or shoots (Morris 1955). The whole branch has these attributes: (a) an equal chance of selection; (b) stability; (c) small enough to be handled to allow collection of enough units to provide an adequate estimate of variance; (d) lends itself to estimating population on a per acre basis; and (e) can be collected without serious loss or disturbance of the insect population.

Most sampling and population dynamics work in Western United States has been done on Douglas-fir. This species is more abundant on a variety of exposures and sites and has a greater ability to endure budworm feeding over a period of years than other host species. It also is valued for economic reasons (Carolin and Coulter 1959). But restriction of sampling to Douglas-fir has hindered population dynamics work in the West because budworm survival and population densities are related to tree species (Williams and others 1971).

Branches of Douglas-fir, however, are not as symmetrical and are much thicker, heavier, and longer than those of true firs. It is practically impossible, therefore, to obtain enough whole branch samples with a pole pruner to adequately estimate budworm populations within and between trees. Entomologists in the Western United States, therefore, have used whole branches and 15-inch (45 cm) twigs to estimate egg mass densities, and 15-inch twigs (cut from the apical portion of the branch) to estimate western spruce budworm and Douglas-fir tussock moth larval populations in Douglas-fir and true firs (Campbell and others 1982, Carolin and Coulter 1972, Mason 1970, Srivastava and Campbell 1982). The twigs are small enough to provide flexibility of sampling design and are easily collected. The proportion of budworm population on the terminal 15-inch portion of the branch, however, may not be as constant as if the whole branch were used. Larvae will move from the exposed periphery of the branch in a negative response to high temperatures and also will move inward or drop down as twigs are defoliated. The apical portion of the branch defoliates first because most of the current year's foliage (the primary food) is located there.

Variation in estimates of larval populations was studied on Douglas-fir whole branches and 15-inch twigs to determine relative efficiency of the two units (Carolin and Coulter 1959). Two whole branches were removed from the middle of the lower crown half from each of 10 trees and four 15-inch twigs were removed by pole pruner from the lower crown half from each of 25 trees. The 15-inch twig samples were considerably more efficient, and took 2.5 to 3 times less work to obtain an accurate assessment than did whole branches.

It is difficult, however, to cut an exact 15-inch branch or twig in the upper crown of a tree with a fully extended 30 to 35 ft (9 to 10.7 m) pole pruner. The length of cut branches or twigs usually varies from 10 to 25 inches (25 to 64 cm). The problem is solved by measuring length and width of all sample branches and dividing the product by 2, and by counting the number of new shoots or buds and expressing the population in terms of food supply; that is, number of insects per 1000 inches^2 of foliage (or per square meter of foliage), or number of insects per 100 buds, or both. These procedures put data on a continuous variable basis rather than on a discrete variable basis.

Egg Masses

Decisions to control insect populations are based partly on their known ability to damage trees, current health of trees, target pest insect population trends, and forecasts of additional effects on the trees. It is imperative, therefore, to have some indication of what the population density of the pest would be the next spring. Weather analysis of the previous season (Hard and others 1980), supplemented by an estimate of density of egg masses produced by the previous generation in critical areas, has predictive value for the early larval density of the next generation. For budworm and tussock moth, egg masses are usually sampled in late summer or early fall to forecast population trends. Timing of sampling at any locality is aimed at an approximate 90 percent egg hatch to ensure the deposition of most eggs before sampling (Carolin and Coulter 1972).

Sampling Procedures—Sample trees similar in appearance to those used to estimate larval populations are selected at various locations within spray areas. The following procedure applies to each sample tree:

1. Use 30-foot extension pole pruners with baskets attached, to cut off the entire foliated portion of branches. Cut one foliated branch from the upper crown, two from the middle crown, and one from the lower crown of each tree to conform to the distribution of egg masses and foliage for most trees (Carolin and Coulter 1959).

2. After cutting, place each branch sample into a polyethylene bag, a tightly sealed paper bag, or a cotton bag. Properly identify the sample by a card that lists area, plot number, and tree number. A large burlap sack conveniently holds 10 to 15 bags that enclose the samples. Place the bags in the shade during sampling to prevent an oven or sauna effect that can occur if eggs or larvae are stored in paper or polyethylene bags exposed to direct sunlight. Heat and humidity buildups are less of a problem when cotton bags are used because the cotton fabric allows limited air circulation.

3. The samples should be held in a walk-in cooler at 45° to 50 °F (7° to 10 °C) until processed.

To process samples, the foliated length and width of each branch sample are measured and recorded and needles with egg masses are removed for examination.

Foliage measurements—Length and width of the foliated branch is measured and area of foliage calculated by multiplying the two measurements and dividing the product by 2. The foliated branchlets are cut from the main branch with hand clippers, and are cut into twigs of convenient handling sizes.

Overlapping grids are drawn on a large cloth with the number of square feet or square meters for each combination of grid dimensions. Branchlets are laid on the grid cloth in a layer spread evenly with no open spaces. Usually only one-quarter to one-half of the cloth is covered. The dimensions of the covered portion of the cloth are recorded in square feet or meters.

To further process the samples, needles of each twig are closely examined for budworm egg masses. All needles with egg masses and other insect material are removed from twigs with tweezers by technicians and placed in a petri dish containing a card that identifies the sample. The contents of each petri dish are examined by an entomologist. Budworm egg masses of each sample are removed and placed in a pill box labeled for sample identification.

Egg masses can generally be separated into "new"—those laid the current year—and "old"—those laid in past years *(fig. 3)*. A combination of characteristics are helpful for separating spruce budworm egg masses *(table 2)*. The separation of new from old egg masses, on the basis of these characteristics is not foolproof, however. Generally, most separations will be correct. A certain number of new egg masses, however, may be called "old" or old called "new." Counts of old and new egg masses are compared to provide trend information on population levels for 2 years. If the total number of new egg masses is substantially less than the total number of old egg masses, chemical control may not be necessary. In addition, the total number of new egg masses may have predictive value on the subsequent defoliation of Douglas-fir in many western coniferous forests (Bullard and Young 1980).

Parasites

Spruce budworm egg masses are parasitized almost entirely by one species of chalcid: *Trichogramma minitum* (Riley). Once this tiny wasp invades a budworm egg mass, it usually parasitizes every egg in the mass. The parasitized egg masses are easily recognized by their black color. If a high proportion of new egg masses are parasitized, a control project may not be necessary. Estimates of the small larval budworm population density in diapause on the trees can be used to determine if control is necessary.

Cocoons of budworm parasites, particularly *Apantales fumiferanae* (Vier.) and *Glypta fumiferanae* (Vier.), are usually found associated with spruce budworm egg masses on needles. Adults of these species attack 1st and 2d instar budworms. The parasite larvae overwinter in diapausing budworm larvae and subsequently emerge from late instar budworm larvae, killing them in the process. Cocoons found on foliage in fall during sampling of egg masses are mainly those of parasites that emerged during early summer of the current year. Because of weathering, few parasite cocoons remain attached to foliage beyond 1 year. Cocoons per 1000 inches^2 of foliage provide an estimate of parasite density during early summer. These parasites presumably are available to attack 1st and 2d instars of the larval population that hatch from the new egg masses laid that summer.

Table 2—*Characteristics for distinguishing between new and old spruce budworm egg masses*

Characteristics	Egg masses	
	New	Old
Color of larval head capsule on unhatched egg	Brown to black	Yellow to light brown
Larval exit holes in eggs	Circular exit holes	Gaping exit holes
Brightness of egg mass	Shiny and clear	Opaque
Dirt particles	None	Tiny, black specks
Stains	None	Present
Color and condition of eggs containing *Trichogramma*[1]	Black, no exit holes	Brown, exit holes
Color of budworm moth's wing scales on top of eggs	Dark	Bleached
Color of specks of yolk	Orange yellow	Bleached
Egg mass adherence to old needles	Sticks well	Falls off easily
Shrinkage of needles	Shrink	Do not shrink

[1] A hymenopteran parasite of budworm eggs.

Overwintering Populations

When field tests and suppression programs are planned, a survey is desirable in the spring to determine if budworm population losses from fall dispersal and overwintering mortality have made direct control unnecessary, reduced the suitability of the area for field tests, or both. The survey is particularly needed if winter was severe. If intense storms occurred earlier than usual, high overwintering mortality in the budworm population could result.

The significance of sampling overwintering western budworm populations is supported by data that show that the density of larvae hibernating on the boles and branches to be a reliable index of the numbers of feeding larvae (Carolin 1950, Carolin and Coulter 1972). A small sampling study of larvae hibernating at midcrown boles and limb sections on nine Douglas-fir trees showed that limb sections were the best sample units for predicting size of western spruce budworm larval populations feeding in the spring on the same trees (Carolin and Coulter 1972). About 79 percent of the larvae at midcrown were found on the limb components. Four regression equations predicting the number

of feeding larvae and mined needles on the basis of limb section data are included in the report. Carolin and Coulter (1972) caution that their results apply specifically to the Blue Mountains of Oregon and those stands in which Douglas-fir trees develop roughened bark on limbs and boles at a relatively early age. Other larger sampling studies are needed to determine the predictive value of estimates of hibernating larvae for feeding larvae on Douglas-fir and other hosts in the geographic range of the western spruce budworm.

The distribution of hibernating larvae on Douglas-fir in Colorado was studied by McKnight (1969). He concluded that the lower crown is a suitable index for estimating hibernating larvae of the western budworm on Douglas-fir in the Rocky Mountains. Whole branch samples can be cut from the lower portion of the tree crown without felling the tree or preventing its use and data recorded for later correlation with numbers of feeding larvae and defoliation. Unfortunately, no sampling design was provided on the basis of the study. Although it may not be necessary to quantify the overwintering population on a per unit basis—simply to determine if enough population survived the winter—some sampling design is necessary to correlate the overwintering population with feeding population and defoliation.

Bole sections cut from the midcrown of 5- to 8-inch-diameter Douglas-fir trees were used to estimate overwintering western budworm populations in the Northern Rocky Mountain Region (Terrell 1959). Such bole sections or billets are cut about 3 ft (1 m) below the junction of smooth and rough bark. They are 15 to 17 inches (38 to 43 cm) long and have a bark surface of 1.35 to 2.50 ft^2 (0.41 to 0.76 m^2) that can be measured easily. Only one billet is taken per tree. Five bole sections per sampling point gave less variable results at each point than did the use of fewer sections. Although Terrell (1959) did not study the distribution of hibernating western budworm larvae on Douglas-fir in the Northern Rocky Mountain Region, he compared larval emergence from overwintering sites on limb and bole samples. An average of 2.9 larvae per square foot emerged from limb samples and 58.0 larvae per square foot emerged from the bole sections of the same trees.

The number of hibernating larvae on Douglas-fir was directly related to the roughness of bark, large bark scales, and lichens (Carolin and Coulter 1972, McKnight 1969). The bark of grand fir and subalpine fir is much smoother than that of Douglas-fir of comparable age and size, and studies are needed to determine the distribution of hibernating larvae on limbs and boles of these true firs and Douglas-fir throughout the mountainous West. The studies are necessary to develop sampling methods that are biologically and statistically sound to estimate hibernating population of the western spruce budworm on its major tree hosts. The study procedures described by Terrell (1959) and McKnight (1969) or the one described in this guide could be the basis for obtaining the needed information on the distribution of hibernating larvae on the various tree hosts of the western spruce budworm.

The best time for sampling is when the probability of late winter storms is low. For many locations within the geographic range of the western spruce budworm, this period occurs between late March and early April. Fall sampling and storing of infested material until spring (before forcing larval emergence) results in a substantially lower (fourfold to fivefold) estimate of overwintering populations (Terrell 1959).

Selection of sample trees will usually be restricted to the most accessible areas because of weather conditions. Nevertheless, the sampling party may need snow machines, snowshoes, skis or all three to obtain the samples. Because of high intertree variance for budworm populations, we believe it best to sample as many points as possible in each drainage or other geographical stratum that will be treated. We also believe it is necessary to include at least two host tree species of the budworm whenever more than one is present within the proposed spray area. The trees should be the same type as those normally selected as sample trees to estimate pre- and postspray larval population densities: about 25 to 50 ft (7 to 15 m) tall and not overtopped or heavily shaded by adjacent trees.

Several sample trees are felled at each sampling point and 4 whole branches taken randomly from each crown third—a total of 12 whole branches per tree. Branches are tied in a bundle with cord and a tag is attached to identify the tree, the sampling point, and the area. The bole of the tree is cut into 2-ft (0.6 m) sections. Every other 2-ft bolt within the crown, and to 6 ft (2 m) below the crown, should be brought in from the field. If the bole within the crown is 20 ft (6.1m) long, then five 2-ft bolts will be taken from within the crown and two 2-ft bolts from the portion of the bolt immediately below the crown, giving a total of seven 2-ft bolts for that sample tree. Each bolt is marked to identify tree and area, and its former position in the tree. The bolt from the topmost position in the crown will be number 1 and the bolt from the bottom position will be number 7.

Branches and bolts from the sample trees are brought into the laboratory or a rearing room and sealed in cardboard containers. Small emerging 2d instar budworm larvae have a strong tendency to seek light and could easily escape if the box is not sealed tightly. A large glass vial inserted into each cardboard container will provide a means to entrap emerging larvae for counting because they are photopositive. Each 2-ft (0.6 m) bolt should be placed into a separate cardboard container. Four branches from the lower crown third are placed in one container, those branches from the middle crown third in another container, and so on. Samples from each tree with a 20-ft (6.1 m) crown, therefore, are contained in 10 cardboard boxes (seven bolt samples, three sets of branch samples).

Second instar budworms usually break diapause and begin to emerge in about 3 to 4 days in a well-heated room with an average temperature of 75 ° to 80 °F (24 ° to 27 °C). They can be easily collected and counted through use of the glass vials inserted and tightly sealed into the cardboard containers. After a week of emerging it should be obvious

if sufficient budworm larvae survived the winter to warrant a control operation. A sample of budworms (3d and 4th instars) taken the next spring on trees on the same plot that provided the hibernating larvae samples can provide the data necessary to determine if overwintering survival on the various tree components is related to total population in the spring.

Distribution of the overwintering population on the trees should be apparent after larval emergence data is analyzed. Analysis of variance (nested) may indicate the distribution and variance of various densities of hibernating larvae among sections within the tree, by tree species, sampling point, and area. Significant differences in population distribution are represented by different value coefficients for tree species, crown size, and other variables in equations that describe the distribution of the hibernating population. Thereafter, those portions of the tree containing the most constant and greatest proportion of large or dense overwintering populations are brought in from the field the next time samples need to be taken. Alternatively, a weighted tree sampling procedure could be designed with the variance component technique.

Sampling hibernating larvae to prevent unnecessary spraying requires considerable labor for often questionable results (Carolin and Coulter 1972). Most sampling efforts have used inadequate sampling designs, and methods of forcing the emergence of hibernating larvae from diapause can result in large errors if the rearing containers are not tightly sealed. Other techniques for sampling hibernating spruce budworm larvae based on washing larvae from their hibernacula have been described (Miller and McDougall 1968, Miller and others 1971). So far we have not seen any published results of these techniques for western spruce budworm, nor do we know their predictive value for estimating populations of feeding larvae.

LARVAL DEVELOPMENT

Spray Applications

Timing of insecticide application and, in a large measure, the success of the control operation, depends upon the correct assessment of spruce budworm larval development rates. Timing is even more crucial when nonpersistent insecticides are used. Most insecticide treatments against the budworm have been applied against 3d to 6th instars. Budworms in these large larval stages cause practically all defoliation and are vulnerable to both stomach and contact insecticides. Insecticides applied against larval instars (3d to 6th) are usually timed so that all larvae have emerged from diapause but none have pupated. If the insecticide is applied while a significant proportion of the larval population is in diapause or is feeding inside tightly closed buds and needles of host trees whose buds have not broken, these larvae may escape any contact with the spray, particularly if the insecticide is nonpersistent.

Insecticides are not as effective against fully developed 6th instar budworm and pupae as they are against young 6th instar and other instars (Robertson and others 1976). If the insecticide is applied after pupation is widespread, a higher percentage of the generation can survive to adults and lay eggs. Consequently, population density of the subsequent generation may be high enough to result in substantial tree mortality and growth loss in forest stands sprayed by the insecticide.

Timing is not as crucial with a persistent insecticide, such as DDT, which has been reported to be available to the target organism for at least 10 days after application (Hurtig and Rayner 1953). Aerial spraying began in most DDT projects at the peak of the 4th instar. Persistence of the DDT on foliage caused it to remain available to the last larvae breaking diapause. Relatively nonpersistent insecticides—pyrethrins, mexacarbate, naled, or other—are applied against the peak of the 5th instar but before noticeable pupation (Williams and others 1978a, Williams and Walton 1968). Spraying was postponed to the 5th instar to increase the probability of contacting all of the larval population—particularly those that break diapause last. Occasionally, however, adequate spraying of nonpersistent insecticides require two applications: an early application timed to affect 3d and 4th instars to achieve early foliage protection; and a late application timed to the peak of the 5th instar to achieve maximum population reduction and protect the remaining foliage *(fig. 4)*.

It is necessary that the criteria for timing the insecticide application be stated clearly in the project work plan; for example, "Spraying will begin within 48 h after 25 (or 50 or 75) percent of the larvae reach 5th instar." Forty-eight hours is enough time to allow for prespray population sampling and other project preparations to be completed, and it is not enough time for substantial numbers of rapidly developing larvae to reach the fully developed 6th instar or pupal stages.

Other insects, such as the Douglas-fir tussock moth, may have extended hatch periods and require different criteria for timing treatment application. To ensure maximum population availability to insecticide exposure before substantial defoliation occurs, applications of relatively persistent insecticides against Douglas-fir tussock moth are timed to begin 4 to 5 days after 70 percent of the tussock moth egg masses begin to hatch. Sometimes two applications 5 to 10 days apart may be needed with nonpersistent insecticides (Williams and others 1978a).

Larval Development Rates

Rates of budworm larval development depend upon several environmental variables of which weather and food quality are two of the most significant. Hot, dry weather while larvae are large instars (3d to 6th) favors high survival and rapid

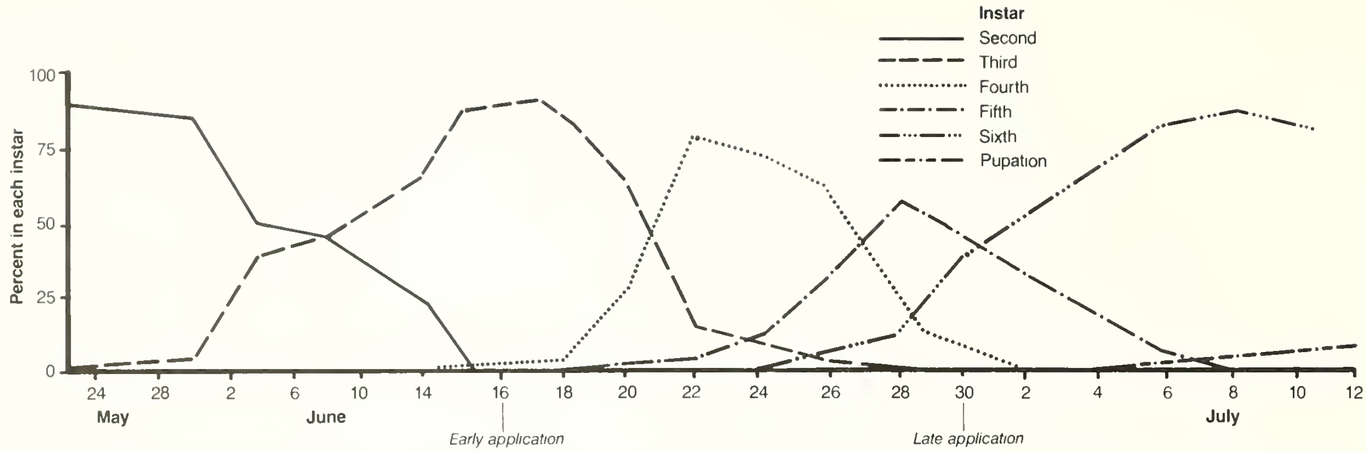


Figure 4—Idealized western budworm larval development following the emergence of the 2d instars from hibernation. An early application of a nonpersistent pesticide would be made when most of the larvae are in the 3d instars. A late application would be timed for the peak of the 5th instars.

development. Increased infestation in an area of northern Idaho and Montana was directly related to hot, dry weather of the previous summer (Hard and others 1980). Larval development rate and survival, however, decrease during cold, rainy weather (Shephard 1959, Morris 1963). Larval development rate is influenced also by the food quality available when diapause is broken. Food quality varies by host tree species and may also depend on host phenology. An extreme example of this situation is provided by comparing survival of spruce budworm on balsam fir with survival on black spruce, *Picea mariana* (Mill) B.S.P.

In the eastern provinces of Canada and in the Northeastern United States, spruce budworm larvae feeding on balsam fir buds develop faster than those feeding on black spruce. This is because of bud phenology and food quality differences between the host tree species (Blais 1957, Swaine and Craighead 1924). Balsam fir buds open approximately when 2d instars break diapause, a time when an ample supply of high quality foliage is available. Black spruce buds, however, open several weeks after larvae break diapause. Larvae on black spruce are forced to mine the hard, less nutritious old needles until buds of the current year open. New black spruce foliage hardens more quickly than foliage of other host tree species and larvae feed on them with difficulty.

In Montana, development rates of Western spruce budworm on true fir—subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) and grand fir (*A. grandis* [Dougl.] Lindl.)—appear to be analogous to those of spruce budworm in the East on balsam fir. True fir buds open about the same time western spruce budworm larvae break diapause. New succulent foliage of true firs appears to be highly nutritious to larvae. Buds on Douglas-fir trees present in the same stand, however, break about 5 to 7 days after those of the true firs (Williams 1968). Variation in time of bud break among Douglas-fir trees at a particular location is large—sometimes 7 to 14 days—compared with the variability shown by true firs—less than 5 days.

On Douglas-fir, budworm larvae are forced to mine hard, less nutritious old needles until the current year's buds break. Once Douglas-fir buds break, however, larvae feeding on the succulent new foliage develop rapidly but do not completely overtake the development of those feeding on grand or subalpine firs. The phenological differences in bud break among host species appear to be responsible for the apparent differences in larval development rate.

Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), and western larch (*Larix occidentalis* [Nutt.]), are also hosts to budworm in the West. Larch buds break earlier than these of any of the host tree species. Buds on Engelmann spruce break after buds on true fir, but before buds on Douglas-fir. Budworm development rates and survival are poor on larch but survival rates on Engelmann spruce are comparable to those on Douglas-fir.

Larval development rates may differ among host tree species at similar locations. It is necessary to keep a record of these rates on the several hosts present in the spray units. In the past, budworm development in Western United States was checked only on Douglas-fir because that species was the most widespread and most commercially valuable. Consequently, many larvae developing on true firs in the spray units were pupae by the spray date. If true firs comprised a significant portion of forest stands in the spray units, a substantial portion of the budworm population was less vulnerable to the insecticide treatment or escaped it altogether. The number of budworm moths available for egg laying in the treated stand, therefore, could be large. Except for some highly localized situations, however, Douglas-fir is overwhelmingly dominant in host western spruce budworm forests. Applications of insecticides timed to budworm development on that species should normally suffice.

If a nonpersistent insecticide is used in the control treatment, it may be necessary to spray twice those areas that have a high complement of true firs and Douglas-fir. The first application should be timed to treat budworm rapidly devel-

oping on the exposed elongating shoots of true firs, recognizing that many budworms on Douglas-fir will be feeding in the still tightly closed buds, or partially closed buds, capped buds, or mining old needles and, therefore, shielded from contact with the insecticide. The second application should be timed to treat budworm developing on Douglas-fir. Peak of the 5th instar could be the target in both situations.

Collection Points

Because the spruce budworm's larval developmental rate depends partly on weather and host tree species, it is necessary to collect larvae from enough locations to cover the range of temperature conditions and host species present in the control area. In a hypothetical V-shaped drainage, for example, larval development sampling or collection points should be established on (a) valley bottoms, (b) midslopes (southern-western exposures), (c) midslopes (northern-eastern exposures), (d) ridgetops (southern-western exposures), and (e) ridgetops (northern eastern exposures).

Significant numbers of grand fir, subalpine fir, or both, are present in moist valleys of some high-elevation areas of Montana, Idaho, and eastern Oregon. Nevertheless, Douglas-fir is the common host tree species throughout forested areas. In this situation, two tree species should be sampled at the sampling point: grand fir (or whichever true fir is predominant), and Douglas-fir. Budworm larvae from each species would be collected and kept separate. Douglas-fir would be the only host species sampled at elevations or slopes where few trees of other host species are present. When recording data, it may be useful to include the locations used to determine larval developmental rates within the sampling framework. The framework is designed according to statistical needs of treatment objectives to estimate budworm larval population densities on foliage. The sampling framework could also provide information on mortality and larval developmental stages.

Sampling need not be done from a single tree of a particular host species but, preferably, from several adjacent trees of the same species or other host species if they are present in substantial numbers.

Two size classes of trees have been used for developmental rate samples:

- Trees 30 to 40 ft (9 to 12 m) tall, with full crown covering at least one-half the length of the bole and not overtopped by larger trees. Larvae are collected from foliage in the top one-half of the tree with a 30 ft (9 m) extendable pole pruner.
- Trees 10 to 15 ft (3 to 4.5 m) tall, with full crown covering almost all the bole, open grown (not overtopped by larger trees). Foliage containing larvae are collected with hand clippers and a net.

The primary criterion is that the tree crowns are open and generally exposed to full sunlight during most of the day. We prefer the shorter trees for sampling ease, and because work can be done more quickly. Developmental samples are processed the same day they are collected. The sampling (collection) points are marked with ribbons and tags and their locations plotted on the project map.

Figure 5—The development rate of spruce budworm larvae in the proposed treatment area must be closely monitored to estimate the date when most of the larvae will be exposed and vulnerable to insecticide treatments.

Sampling Procedure

Sampling collection points are established no later than 1 week after substantial numbers of budworms break diapause. Early development of insects can be watched every 3 to 4 days. If development is rapid, however, insects should be watched more frequently—perhaps every 2 days. It may be possible to estimate probable spraying dates on the basis of current early developmental rates and information of previous years.

Samples should be collected daily starting at approximately 2 weeks after the budworms break diapause and at least 10 days before estimated spray date. Enough buds and developing shoots with signs of current larval feeding are clipped at each sampling point to fill a 1-gal (4.5 l) ice cream carton. Enough foliage is collected to allow examination of 50 to 100 larvae per sampling point or location. Many buds will show signs of larval feeding but are later abandoned by the larvae. If larvae are collected directly from foliage and buds are not dissected, the field developmental rate data will be heavily biased toward larger, easily seen late instars.

Samples can be collected in ice cream cartons and each carton labeled by location, tree species, and area. Cartons from all sampling locations are brought into the laboratory for processing by early afternoon of the day samples are collected.

Buds and developing shoots from each carton are examined thoroughly and dissected. All larvae are removed and placed into a vial or jar containing 70 percent alcohol. The vial is properly labeled to identify the sampling point, tree species, and area. The work must be done carefully. Second and third instars are tiny and easily missed. If substantial numbers of early instar larvae are missed, the results of sampling will not represent actual field conditions and will lead to erroneous conclusions about field developmental rates. This work could be used as training exercises for workers hired to examine foliage from pre- and postspray sampling periods of the proposed field experiment or control job.

Larvae are removed from the vials and examined by an entomologist. The entomologist identifies various instars with the aid of a microscope and counts the number of budworm larvae in each instar. The project entomologist graphs the data that illustrates larval developmental rate at each sampling point *(fig. 5)*. All work should be completed before the project entomologist concludes the work day. During hot weather it is necessary to plot instar development daily to determine precise timing for effective application.

THIRD TO SIXTH INSTARS

Two main procedures for ascertaining effects of various insecticide treatments are hypothesis testing and population estimates. Differences in density estimates of budworm and other target pest populations are compared before and after treatment; mortality estimates are compared between treated and untreated populations (see *Experimental Design*). Estimates of mortality and pretreatment and posttreatment population densities must be as statistically precise as a given level of financing can afford. Statistical precision for pretreatment and posttreatment population estimates is aided by remeasuring populations on the same tree.

Sample Trees

Species—Several budworm host tree species are usually present throughout the insect's geographic range. Because the population densities, survival and larval development rates vary with tree species, it is a mistake to choose sample trees for estimating population densities from just one host tree species in proposed spray areas. Choosing sample trees from the several host trees present in the spray area is particularly important in operational projects. Because of the desire to limit variables, however, restricting sampling to the dominant host tree species may be appropriate in some field experiments. However, the recommended procedure to follow where several budworm hosts are present is the following:

• Divide the proposed spray area delineated on the map into its constituent forest types, for example: Engelmann spruce—subalpine fir, grand-fir—Douglas-fir, Douglas-fir—ponderosa pine, nonhost types.

• Determine acreage for each forest type and use some form of random sampling-tree selection within the stratified host types.

Size—Whenever possible, select trees most representative of the forest canopy for sample trees. In most instances, trees in the dominant position of the forest canopy are too large to adequately sample with a 30-ft extension pole pruner. Of necessity, therefore, the choice is limited to those trees that can be reached with the pole pruner. Sample trees should have open crowns, that is, crowns not overtopped or sheltered by taller adjacent trees. Pre- and postspray population estimates on the basis of sampling small, overtopped trees are sometimes in error because of budworm larvae dropping onto the study trees from taller adjacent trees between sampling periods. It is also desirable to use open-grown trees to obtain an optimal number of buds during sampling. Portions of the crown that are shaded (usually the lower one-third) do not produce many buds. Developing vegetative buds are the natural feeding sites for larvae. Population density of western budworm larvae on open-grown Douglas-fir trees 70 to 80 ft tall do not differ significantly from that of open-grown trees 30 to 40 ft tall (Campbell and others 1982).

Because the primary purpose of our population sampling is to obtain data on effects of insecticide treatments on target pest populations, study trees selected should be those on which target pest populations are most exposed to aerial applications of insecticides. Situations where treatment effects are confounded by variables associated with tree height, stand density, overhead canopy, and other factors that serve to shield populations on study trees from aerially applied insecticides should be minimized.

We are not interested in estimating population densities on representative forest trees in various host forest types. It is on representative forest trees where random sampling and independence of observations are of paramount importance.

Although we use sampling formulas appropriate for random sampling and data distribution described by a normal universe (bell-shaped data distribution) to compute the sampling errors for our estimates, some form of systematic sampling to estimate spruce budworm populations in the forests must be used, because it is too expensive to randomly sample trees in a forest due to limited accessibility and high travel costs. Some system of incorporating random sampling, stratification, and sometimes systematic sampling, is usually developed.

Unless the sample is very small, there is a slight risk in making statements about differences between estimates of sample universe characteristics and true universe characteristics on data from a nonnormal universe (Cochran 1963). The derived universe (consisting of a large number of sample estimates of a particular universe characteristic obtained from a large number of samples) must be distributed normally. If the parent universe deviates even widely from normal, the derived universe may still be near enough to normal to allow the use of the sampling error formulas. This is more true in larger samples.

Plots—After the proposed spray area is stratified into its component host and nonhost types, a number of sample trees are allocated to each host type according to that host type's dominance in the spray area. Actual selection of tree plots is based on accessibility to road systems, helicopter landing sites, navigable streams, and forest trails. Selection of plots dependent upon accessibility introduces another consideration into the sampling effort. We have no evidence, however, that the budworm population or any insect populations are distributed in the forest according to avenues of human accessibility.

Plot locations are randomly selected as soon as possible within the various host-type strata. Number of plots depends upon the sampling effort needed and number of samples deemed adequate to estimate a certain size population within a specified sampling error or precision.

Some forest pest managers believe that area coverage is best achieved by systematic sampling. They select trees systematically with a grid or line transect system over the area. Plots consist of single trees equidistant from each other. This kind of distribution is difficult to obtain in natural forests, particularly in the mountainous West. It could be obtained in plantations, in flat areas of the South, and in upper mid-West areas at a high cost of sampling.

We recommend using the road system and other avenues of accessibility as much as possible to cover the area. Supplement this coverage by several compass lines locating tree plots at specified intervals within the budworm host type.

Sampling Procedure

Prespray sampling of the insect population begins about 48 hours before the spray date and is completed by evening of the day before application begins. The sampling procedure is basically that described by Carolin and Coulter (1972), and Campbell and others (1982). It consists of counting the number of larvae and buds and measuring the foliage area of 15-inch (45 cm) twigs or branches removed from the midcrown of sample trees. Branches are cut with a 30-ft extendable pole pruner from midcrown of the sample tree for prespray larval sampling *(fig. 6)*.

Postspray sampling begins after the insecticide activity has completely ended. This will vary with insecticides, formulations, weather, and surfaces. In the past, postspray sampling began 4 to 5 days after the relatively nonpersistent chemical mexacarbate was applied and 10 days after a more persistent chemical such as DDT was applied. Several postspray samples may be desirable.

Two or more postspray samples of 5th and 6th larval instars provide data for mortality curves and their slopes for treatments and areas. In addition to at least two postspray sampling periods during the 5th and 6th instar stage, it may be desirable to sample the pupal stage and egg masses laid by the treated generations of insects to discern ultimate treatment effects.

Branch samples taken in late summer according to the following procedure provide information on spruce budworm pupae, adults, and new egg masses—all at the same time:

- Some statistically determined number of 15-inch branches are cut from crowns of sample trees for the postspray larval sampling. Data from previous work shows that average budworm population densities occur at midcrown in most trees (Williams and Walton 1968). That crown portion is adequate for prespray sampling. It may be inadequate, however, for providing a large number of postspray samples. It may become necessary, therefore, to use other portions of the crowns to obtain all the postspray samples. Sampling crews must avoid taking lower crown branches that contain no buds.
- Each 15-inch twig is cut and allowed to fall into an 18- to 20-inch (45 to 50 cm) cloth basket attached to the pruner just below the cutting head. The basket catches larvae from branches that drop during cutting. The pole pruner is carefully lowered to avoid hitting other tree branches and to prevent the cut twig from falling out of the cloth basket.
- Samples and larvae are carefully placed into a large polyethylene bag, tough paper sack, or cloth bag along with a card stating study area and date, plot number, tree number and species, and branch number.
- The opening of the sample bag should be closed by twisting it tightly, doubling the twisted end, and retaining it in that position by strong rubber bands. This procedure reduces the opportunities for larval escape.

Sample bags are placed into large burlap sacks (old feed sacks are excellent) and placed in the shade. Usually 10 to 15 bags can be placed in a sack depending on the size of the branches. Samples from the same tree are kept together. This makes it easier to assign specific trees to an insect checker so that the same person examines both pre- and postspray branches. The procedure helps minimize variation in insect counts and branch measurements because of individual checkers.

If a laboratory is used to process branch samples, the samples are taken to the field laboratory and placed overnight in a walk-in cooler. Temperature of the cooler, between 40 ° and 45 °F (4 ° and 7 °C), is enough to reduce activity of the larvae without killing them.

Processing Branch Samples

The branch samples can be processed in the field for Douglas-fir tussock moth larvae *(fig. 6)* or in the laboratory for

A

B

C

Figure 6—The sampling of Douglas-fir tussock moth and spruce budworm larval populations consists of removing twigs or branchlets from host trees by an extendable pole pruner (*A*). These twigs are either examined for insects and measured at the sampling site (*B*) or are placed in bags and transported to the field laboratory (*C*).

budworm larvae *(fig. 7)* to obtain insect and bud counts and branch measurements. Difficulties of obtaining accurate insect counts in the field and laboratory were discussed by Morris (1955). A processing procedure has been developed for use in field laboratories and is accurate enough to replace expensive foliage examination commonly used in the past (DeBoo and others 1973, Martineau and Benoit 1973). This procedure uses a screen table inserted into a galvanized drum held at an angle on a folding wooden stand. The apparatus consists essentially of these five parts:

• A perforated cap (for a 16-oz screw-top widemouthed glass jar) welded near the bottom end of the galvanized drum to fit a 2-inch hole, and a handle fixed near the point of balance of the drum on the opposite side

• A removable rectangular iron screen tray (23.2 by 18 inches, mesh 0.5 inch) framed with a welded steel rod

• A 16-oz collecting jar

• A paint brush (1½ inches wide)

• A folding wooden stand built so as to keep the drum at the required angle and height when in operation and fitting inside the drum during transportation.

Insect larvae are separated from the foliage in three steps: (a) beating the branch sample vigorously against the screen table and side of the drum (30 strokes in all), (b) brushing down the screen and the inside of the drum to direct larvae into the jar, and (c) removing the jar for examination of contents.

Figure 7—Technicians in the field laboratory carefully examine foliage samples for larvae. The buds or new shoots are counted and the foliated areas of the samples are measured.

Foliage samples taken before the peak of 3d instar should be placed in nylon mesh bags and returned to the field laboratory. Here the budworm larvae can be allowed to develop to 3d instar on foliage at room temperature before branch samples are processed in galvanized drums. All other branch samples for pupae and egg masses can be processed at the field laboratory.

Budworm and other insect larvae dislodged from each branch sample are counted and total number for each branch and tree recorded on data forms. Length and width of the foliated portion are measured to the nearest one-half inch. Total number of buds or shoots of the current year are counted, examined for feeding injury or defoliation, and placed into different damage classes. Damage Class I = 1 to 25 percent, II = 26 to 50 percent, III = 51 to 75 percent, and IV = 76 to 100 percent defoliation. Changes in actual shoot counts and percentage of total counts in each category over time help to assess defoliation damage.

For each branch the following information should be recorded:

1. Date and treatment
2. Study plot number
3. Tree number and species
4. Branch number
5. Total number of the target insect

Figure 8—Larvae removed from the foliage samples are counted, and then placed in petri dishes for examination by the supervisory entomologist. A number of these larvae may be fed an artificial diet and reared for possible emergence of parasitoids.

Figure 9—The effects of various suppression programs by insecticides on the parasitoids of forest defoliating insects must be monitored to obtain data necessary for the development of pest management strategies. Such strategies may use a combination of chemicals and biological control agents.

6. Branch length
7. Branch width
8. Total number of buds or shoots
9. Number of undamaged shoots
10. Number of shoots in Damage Class I (1-25 pct defoliation).
11. Number of shoots in Damage Class II (26-50 pct defoliation)
12. Number of shoots in Damage Class III (51-75 pct defoliation)
13. Number of shoots in Damage Class IV (76-100 pct defoliation).

About 100 budworm larvae are collected from each tree cluster or plot during each sampling period and placed into rearing for parasite emergence. The larvae are reared in petri dishes *(fig. 8)*. Several budworm larvae (no more than 10) are placed in each petri dish—in a manner not to injure them—and are fed an artificial medium (Lyon and others 1972). Date, study plot or cluster number, or both, are recorded on each dish. Petri dishes are held at the field laboratory and processed to determine percent parasitism, including identity of the parasite complex. Parasite emergence data will indicate the possible effects the various treatments may have had on hymenopterous parasites that parasitized 2d instars the previous fall. These parasites emerge from 5th and 6th instars and form cocoons. Treatment may also affect tachinids that attack 5th and 6th instar budworms and emerge from budworm pupae *(fig. 9)*. Processing the parasite samples can be done after the peak work periods for the field tests have passed.

We recommend the following procedure for processing pre- and postspray branch samples at the laboratory:

1. Samples are removed from the cooler and sorted according to tree. Tree numbers are assigned to technicians counting the insects so that each person will examine foliage from the same tree for pre- and postspray population counts.

2. The 15-inch sample branches are removed from the polyethylene or paper bags. All insects are removed from the branches and bags and placed into petri dishes by technicians supervised by an entomologist *(fig. 8)*. It is necessary to have enough technicians (8 or 10) so samples can be processed rapidly. With rapid processing, storage of samples is minimal and the probability of significant budworm larval mortality or escape during processing is reduced.

3. After insects are placed in petri dishes, those examining the foliage measure the longest length and width of each sample branch and count the number of current year's buds (including, of course, developing foliage shoots). Total number of buds or shoots of the current year are examined for feeding damage and placed into one of the four damage categories previously described. Data for bud counts, branch measurements, and damage classification are recorded on the card taken from each bag identifying the sample.

4. The identification card is affixed to the petri dish (dishes) containing the insects removed from the branch samples. The dishes are given to the supervising entomologist.

5. Budworm larvae are separated from other insects and counted. Insects found associated with the budworm are identified and counted. Two identification keys have been useful in separating larvae of associates of the western spruce budworm on new foliage of Douglas-fir and true firs. One (Carolin and Stevens 1979) separates small larvae in opening buds and new shoots, and the other (Carolin and Stevens 1981) separates advanced instars on expanded foliage of Douglas-fir and true firs.

6. Tree and branch numbers, bud counts, sample branch measurements, damage counts, and budworm and associate insect counts are recorded on an appropriate data form, and larvae are placed in petri dishes containing food medium for parasite rearing *(fig. 8)*.

INSECTICIDE APPLICATION

The task that perhaps influences success or failure of field tests and control programs most is field application of the insecticides. Efficacy and safety of insecticide formulations developed in the laboratory ultimately are determined in the field. Field effectiveness of formulations against target insects are influenced by many more variables than those affecting the organisms in laboratory bioassay. Efficacy is determined not only by properties of the chemical formulation and dosage applied, but also by a complex system of other variables that affect coverage of the treated areas and deposit on target organisms. These include types of aircraft, application equipment and techniques, temperature, rain, sunlight, wind turbulence above and within the forest canopy, spray physics, and forest stand density and structure. Insecticides must be applied in a manner to maximize coverage of the target area and enhance the probability of impingement of the insecticide droplet on the target insects.

Equipment

Equipment designed to meet job specifications and properly calibrated is required. If this equipment is commercially available, owners or lessors must be sought out and encouraged to bid on the advertized or contemplated control operations or field tests.

The aircraft (airplane or helicopter) must be powerful enough to carry a full load of spray from the airport (heliport) to the spray blocks, spend sufficient time for orientation, apply treatment, and return with an adequate reserve of fuel. The aircraft must be capable of carrying the full load with a safe margin of fuel at all elevations and must have the desired spray system attached with adequate power to run it.

The spray boom should be made of a strong, lightweight material and be large enough to allow easy flow of the insecticide formulation under pressure. Nozzles should be designed to produce a restricted range of spray droplet sizes without leakage.

The aircraft and spray system must be calibrated before they are accepted and used on the tests or operations. Calibration of hydraulic pump systems generally consists of setting up the entire spray system on the aircraft, and spraying the insecticide-dye mixture and running it while on the ground *(fig. 10)*. At this time all nozzles should be checked to see if (a) they are new, (b) they are applying the spray uniformly at the desired pressure, and (c) they (or other parts of the

Figure 10—The spray equipment and system attached to the aircraft must be inspected to see that it is in proper working order and calibrated to deliver the spray formulation at the desired volume rate and droplet size over the target area.

spray system) are not leaking or drooling. Small fixed-wing aircraft with spray systems operated by a wind-driven pump must be flown for calibration.

Once the spray system is determined to be satisfactory for operational conditions, the aircraft should be required to make at least three passes over a line of Kromekote or other deposit cards to check spray pattern at sunrise or just after sunset during a period of little or no wind. Cards are placed 10 ft apart in a line 200 ft long at 90 degrees perpendicular to the direction of the prevailing wind. The aircraft is required to fly upwind over the center of the line at 50 ft above the ground while spraying the insecticide dye mixture. Cards are then checked to determine overall swath width of the aircraft and spray system and the working swath of the aircraft. The working swath is defined as that part of the swath at which the deposit is more than over 20 percent of the volume applied (usually 0.2 gal/acre [1.9 l/ha]). To compute the 0.2 gal per acre, the cards from the calibration test are compared to standard cards with drops of the same size and similar colored dye. The cross-swath check also can be used to determine if the swath pattern is uniform. Areas of very low deposit at swath center indicate nonfunctioning or misoriented nozzles on the aircraft. If such skips are found, the aircraft contractor is required to change or rearrange the nozzles so that a uniform swath is obtained.

A method for determining optimum swath widths and spray application rates to obtain a uniform spray coverage of 20 to 25 drops per square centimeter at the top of the forest canopy was described by Dumbauld and others (1980). A computerized atmospheric dispersion and canopy-penetration model is used to demonstrate the method under various meteorological conditions for five different types of spray aircraft.

Formulation

Usually, some mixing of the formulation is necessary before it is loaded into spray tanks. A small quantity of dye (such as Rhodamine B Extra base) or tracer is added to aid in assessing deposit. Some formulations must be diluted with a carrier material. Also, some dilutions are not stable mixtures and will settle out if allowed to stand for more than a few hours. Once settled, some mixtures may not resuspend. It is necessary, therefore, to know exactly how much will be needed for each day's work and mix that amount just before treatment. If help is desirable, a representative of the insecticide manufacturer is usually available and can be contacted for consultation.

Figure 11—The spray formulation must be properly mixed and free of dirt and other contaminants that would plug the spray system of the aircraft.

Mixing of the formulation preferably should be done in a 500-gal or smaller mixing tank *(fig. 11)*. A suggested sequence of adding and mixing of ingredients is as follows:

1. Determine exact amounts of formulation that will be used each day.
2. Check the mixing tank and hoses to see that they are clean and contain no sediment or water.
3. Place the diluent (usually no. 2 fuel oil or water) into the mixing tank.
4. Add the dye or tracers to the mixture in the mixing tank.
5. Mix thoroughly for 10 minutes (this mixture can be formulated and stored the day before).
6. Add the insecticide formulation from the manufacturer to the mixture in the tank.
7. Mix thoroughly for 30 minutes.
8. Mix again for 15 minutes each hour while loading the formulation into the aircraft.
9. Set aside some tank samples for bioassay and laboratory analyses.
10. Do not attempt to save and reuse any formulation not used the day it was mixed unless it is a solution that does not separate over time. Place any extra formulation in the drum for disposal.

Spray Application

Boundaries of the treatment area are identified with markers that are readily visible to pilots. Corners may be marked with fluorescent streamers located on the top of the dominant tree nearest to each plot corner. Streamers can be positioned in the top of a tree by shooting a line over the tree with a crossbow *(figs. 12, 13)* or line guns and pulling the streamer into the tree's crown (U.S. Department of Agriculture, Forest Service 1974). The corner markers must be conspicuous from the air to assist spray pilots with locating plots and to provide reference points to help in gauging spray width *(fig. 12)*. Proper location of highly visible markers increases the accuracy, efficiency, and safety of aerial application. Several methods for placing colored markers in and above the treetops so that they are easily visible from the air to spray pilots have been described (Maksymiuk 1975).

Before applying the insecticide, the pilot must be oriented to the spray area boundaries, the working swath widths of the aircraft, and dangerous topographic features *(fig. 14)*. The pilot and the aerial observer must plan flight lines for applying the insecticide. Because most spraying is done in early morning, flight lines are oriented so that the pilot can avoid facing

A

B

Figure 12—Fluorescent streamers located on the tops of dominant trees (*A*) or in the open (*B*) are highly visible to spray pilots and help mark plot corners and boundaries. Proper location of these streamers increase the accuracy, efficiency, and safety of aerial application of insecticides.

the sun. If plots to be treated are small, 20 acres (8 ha) or less, it may be possible for an aerial observer to guide the pilot on spray runs. This guidance is best achieved if the observer is in another aircraft flying above the spray aircraft. In this position the aerial observer can see how evenly the spray coverage is applied, can pick out uneven swathing and missed spots, and can guide the pilot to obtain desired coverage. Each spray swath should be marked on a black-and-white aerial photo. The application rate in acres per minute and the number of acres treated can be calculated by knowing

A B

Figure 13—Fluorescent streamers can be positioned in the tops of trees by using crossbows (*A*) or signal guns (*B*) to shoot lines with attached streamers over the trees.

Figure 14—Project supervisor must orient pilots and aerial observers to the spray area boundaries and dangerous terrain features of the area. Flight lines must be planned to insure proper spacing of the working swath widths of the spray aircrafts.

the working swath width, swath length, and speed of the aircraft (Barry 1978).

While applying the insecticide, the aircraft is required to fly at a height of 50 ft above treetops at a designated speed and swath width.

Spraying should be prohibited when any of the following conditions exist:

- Wind velocity exceeds 6 mph, and an upslope wind pattern develops
- Temperature exceeds 70 °F (20 °C)
- Snow, water, or ice covers the foliage
- Rain is predicted to fall within 6 hours
- Fog covers part of the area to be treated or the flight path of the aircraft between the airport (heliport) and the test site.

Meteorological conditions should be monitored in each spray area before and during spray operations. A meteorological station should be set up at least 1 week before spraying to determine wind patterns over the spray area during the

day. Temperature, relative humidity, wind velocity and direction, cloud conditions, and presence or absence of an inversion layer should be measured in the vicinity of the spray site (figs. 15, 16, 17). An inversion layer can be detected by means of a wire sound weather balloon, inflated with helium, which can be used to record temperature at 10-ft intervals above ground level to an elevation of 150 ft (fig. 17). Inversion layers form an impenetrable barrier for small spray droplets. The layers begin to break up at sunrise, and mild turbulence results, which aids the distribution of the small droplets within the target area. Meteorological data are collected at the start, midpoint, and end of spraying (Barry and others 1978).

Figure 15—Smoke released in the spray plot can indicate wind speed and direction within and above the forest canopy. This knowledge allows the spray pilot to compensate for wind drift during his spray runs.

Figure 16—The various meteorological parameters of the spray area should be monitored before as well as during the actual spray application. Wind patterns over the spray area should be known for different periods of the day.

Figure 17—Inversion layers over the spray areas can be detected and measured with a wire sound weather balloon inflated with helium. It can take vertical temperature profiles at 10-foot intervals from ground level to 150 feet.

SPRAY COVERAGE

Barry and others (1978) have described the equipment and materials, and field and laboratory procedures for assessing insecticidal sprays released over forests. Conversion and computation tables, spread factor equations for some formulations, and sample forms are found in their reference source and training guide.

An assessment of spray coverage helps to determine the quality of the application to the target area. It indicates whether the spray equipment was working properly and if the spray was applied evenly to the target area. It shows gaps in the coverage and indicates areas that were missed. It can also show spray contamination in nontarget areas if those areas are sampled. Rapid assessment of spray coverage can correct deficiencies in application and, thereby, improve quality.

Assessment of spray deposit provides information on the physical characteristics of the spray such as droplet size, droplet density, and spray mass. These characteristics are sometimes related directly to mortality of the target insect, and inversely, to various indices of injury to the host. The best way to evaluate spray effectiveness on defoliating insects is to directly examine spray deposits on the target insect or its substrate—the tree foliage. Usually these assessment methods are tedious and time-consuming, and their practicality is limited to research. Spray deposit in aerial treatment of forests in experiments, and pilot and operational control projects are usually assessed with paper cards, aluminum plates, or both.

Spray Deposit Cards

Oil-sensitive cards have been used in control projects to help evaluate the spray coverage over the treated area. Because the solvent or insecticide carrier usually has an oil base (for example, cycle oil), spray droplets 40 μ and larger register on oil-sensitive cards. The cards are placed around the sample trees and at specified intervals along lines laid out perpendicular to the airplane swath widths. The quantity of spray on a card can be roughly estimated by visually matching the card with standard cards carrying known amounts of deposit (Davis 1954).

White Kromekote cards are made from a cast-coated highly calendered stock with a finish that shows droplets with sharp, distinct edges (Markin 1978; *fig. 18*). A spray droplet impinging on these cards spreads out and forms a circular stain or spot the size of which is related to the size of the droplet. This relationship, called spread factor, is the ratio of the diameter of the stain to the diameter of the drop causing it (Waite 1978a). The size of the droplets forming the spot on the card can be calculated by measuring the diameters of the spots on the card and knowing the spread factor of the spray. Spread factors on white Kromekote cards differ among pesticidal formulations but many have been calculated (Waite 1978a).

Dyes that increase the visibility of spray droplets are usually added to insecticide formulations to help register droplets on cards, foliage and insects. These include fluorescent dyes such as water soluble Brilliant Sulpho Flavine FFA (BSF), Rhodamine B extra S, and oil soluble Rhodamine B extra base. Nonfluorescent dyes include water soluble Nigrosine and oil soluble Sudan Deep Black. Although analytical methods that use soluble fluorescent dyes are more sensitive than those that use soluble nonfluorescent dyes, fluorescent dyes generally fade rapidly in intense sunlight (Himel and others 1965). Also, only fluorescent dyes have been used on trees and insects, and they are absorbed in an irregular, unpredictable manner by both foliage and insects.

Spectral counts of droplet stains on Kromekote cards are used to determine the deposit characteristics of spray dissemination systems on the spray aircraft and also the quality of

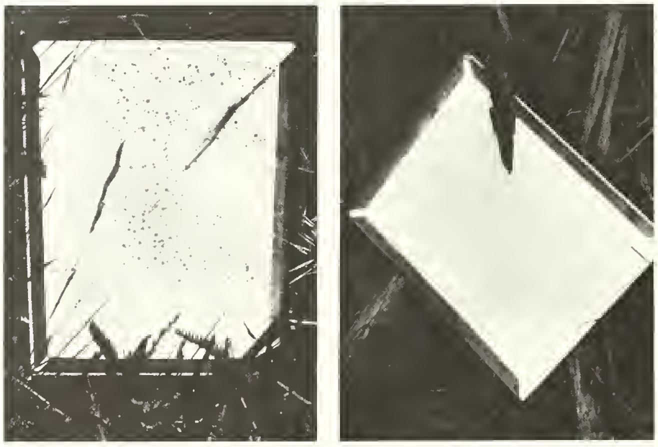


Figure 18—Spray deposit cards can indicate the spray coverage, the spray droplet spectrum, and the volume of insecticide deposited on the target area. Without this information it would be difficult to evaluate the quality of the spray application job.

the aerial application of the insecticide. Procedures for making the spectral counts have been described (Dumbauld 1980; Dumbauld and Rafferty 1977, 1978; Dumbauld and others 1980; Maksymiuk 1978).

Spray droplet size registered on cards aids in determining spray atomization (Maksymiuk 1964, 1978). The degree of spray atomization can significantly assist the effectiveness of aerial spraying because it influences spray coverage through distribution, evaporation, and drift. The degree of atomization formerly expressed as mass median diameter, is now expressed as volume median diameter (VMD). This is the drop diameter that approximately divides the spray volume into two equal parts—50 percent of the volume is in droplet sizes above the VMD and 50 percent in droplets below it (Maksymiuk 1964, 1978). The VMD is determined from samples of the drop size spectrum visible on the cards. Consequently, any inferences from sampling apply only to the droplet spectrum above 40 μ in diameter.

Density of the spray droplets on the cards indicates the quality of insecticide coverage and, therefore, if a forest has been treated satisfactorily. The amount of active ingredient deposited is related directly to the number and size of the drops being deposited. The number and size range of spray droplets that impact on spray cards and plates are highly variable and are largely influenced by the volume and size range of the droplets in the material that is applied, by the location of the cards, the size of the forest openings, density of the forest canopy, wind speed and direction, and the time of day. Nevertheless, pest control specialists have generally assumed that mean droplet density, or the amount of deposit in gallons per acre, estimated from spray cards placed near sample trees and in open areas in the forest is a reliable index of insect mortality. Several studies have reported good correlation between spray droplet densities and volumes of deposit on the cards, and insect mortality in the area (Flavell and others 1977, Hard and others 1979, Johnson 1963, Young 1978). In other studies, little or no correlation was found between insect mortality and spray deposit (Buffam 1965, Maksymiuk 1963b).

One study showed that the portion of the spray droplet spectrum below 50 μ resulted in 97 percent mortality of the budworm populations treated with mexacarbate (Himel and Moore 1967). Because few droplets below 40 μ are visible on spray deposit cards (Thornton and Davis 1956), the most effective portion of the droplet spectrum could not be detected. High correlations between number of spray droplets and budworm mortality were obtained when the unseen portion of the droplet spectrum (drops 40 μ) accompanied the visible portion of the droplet spectrum down to the target. At those times when the correlation was low, small unseen droplets evidently did not accompany the larger visible droplets to the target, but fell elsewhere.

Despite difficulties in recording the presence of the small droplets and their reliability in predicting insect mortality, spray cards are useful in indicating areas that may have been missed or that were inadequately covered by the spray. The cards are easy to use and are the most practical sampling surface for experimental, pilot, or operational control projects.

Fluorescent Particles

Fluorescent particles aid field experiments because deposit can be estimated by counting particles on insect larvae and foliage with an ultraviolet light source and a stereomicroscope. Counting particles on larvae and foliage, however, is time-consuming and, therefore, impractical in pilot and operational projects. Insoluble fluorescent particles were used experimentally in western Montana to determine size and number of aerial spray droplets impinging on spruce budworm larvae (Himel and Moore 1967). A sample of 1113 larvae affected by the spray was microscopically examined with ultraviolet light. Fluorescent particles on each larva were counted to determine the size of the largest droplet. Because the number of the micron-sized fluorescent particles and volume of insecticide spray liquid were known, the number of fluorescent particles in each spray droplet was a direct measure of droplet size (Himel and others 1965).

Ninety-three percent of the spruce budworm larvae had no drops larger than 50 μ in diameter (Himel and Moore 1967). Distribution of droplet size was examined on a subsample of 346 larvae. Ninety-seven percent of the total droplets on these larvae were below 46 μ. Because 95 percent of the total volume of most conventional sprays used in spruce budworm control consists of droplets larger than 50 μ in diameter, only 5 percent of the insecticide applied to the forests was effective in killing spruce budworm larvae from direct aerial contact (Himel and Moore 1967). These conclusions have been supported by other studies (Barry and others 1977).

Fluorescent particles offer improvement over spray deposit cards. Larvae and foliage from various plants in the treated area can be illuminated by ultraviolet light, examined under a microscope, and the number of fluorescent particles counted to determine spray atomization in insecticide coverage. Droplets smaller than 21 μ, however, have a high probability of having no fluorescent particles and, therefore, are not visible (Himel and Moore 1967).

Foliage

Because tree foliage intercepts much of the spray and is the actual habitat of target insects, it should provide a better index of the amount of insecticide actually available to them than cards and plates. Although deposit data from foliage is more expensive to obtain, it may be more closely related to insect mortality than data from cards and plates. Foliage samples are being used increasingly in research to assess spray coverage and deposit (Maksymiuk and others 1975, Barry and others 1977, Williams and others 1978a, Barry 1978). Data on the amount of spray deposit, spray droplet sizes, densities, and quality of coverage can now be obtained from foliage. The amount of spray deposit is estimated by spec-

trofluorometric analysis. Spray deposit sizes and density data are obtained by examining foliage under a dissecting microscope equipped with a reticle (Barry and Ekblad 1978).

Several criteria must be met when assessing foliage (Barry 1978):

• Dye concentrations must provide suitable contrast.

• The droplet spread factor on target foliage must be determined.

• The sensitivity and detection threshold of the magnifying instrument used must be known.

• Droplets must dry in a reasonable time without excessive running.

Because foliage assessment methods are time-consuming and costly, various investigators have sought relationships between insect mortality and spray deposit densities on foliage and on Kromekote cards placed on the ground (Barry and others 1977, Maksymiuk 1963a, Maksymiuk and others 1975). To date, we have no evidence that foliage assessment methods provide better data on spray coverage of the target area or are better predictors of insect mortality than Kromekote cards and aluminum plates.

Volumetric Methods

Many investigators have not been satisfied with the quality of information obtained from spray cards or with the limitations of tracer materials such as fluorescent particles. Greater research effort has been expended on volumetric methods to estimate the number of gallons per acre present on the target area. Currently, much of this information is being obtained from tree foliage and aluminum plates, which are washed and the resulting solutions analyzed with a fluorometer (Yates and Akesson 1963). The amount of dye in the wash solution is compared with the amount originally added to the spray and, given the area of the plates, the volume of spray deposited, in gallons per acre, can be calculated.

Large variations in spray deposit and larval counts have been found between trees because of (a) screening effects of adjacent trees on spray deposited on foliage and on plates and cards placed on the ground near sample trees (Maksymiuk 1963a) and (b) natural variation of larval distribution within and between trees compounded by sampling error. Consequently, when amount of deposit and percent larval mortality were analyzed on a tree-by-tree basis, correlation between spray deposit and larval mortality was low (Maksymiuk and others 1975).

High correlations were obtained between amount of deposit on tree foliage and larval mortality when sample trees were grouped in various categories or classes on the basis of mean spray deposit recovered from trees (Maksymiuk and others 1975) or when sampling of larvae and spray deposit on cards were based on tree clusters (Flavell and others 1977).

Recently, high correlations between western spruce budworm and Douglas-fir tussock moth larval mortality and volume deposits, estimated from spray cards and aluminum plates placed beneath sample trees, were obtained on a tree-by-tree basis. Volume deposit estimates for three data sets were used to estimate dosage to which the insects were actually exposed (Williams and Robertson 1983). The estimated dosage variable provides an interface for use with the logic of a laboratory efficacy model that calculates expected mortality (Force and others 1982). For the three data sets examined, prediction success (as measured by a χ^2 goodness-of-fit test) ranged from 73 to 95 percent (Williams and Robertson 1983). And mean mortality predicted or calculated by the model was within 5 percent of that actually observed in all situations.

Spray Deposit Samples

The spray deposit assessment methods used in the 1972–1974 Douglas-fir—tussock moth insecticide program were foliage, aluminum plate, and spray card sampling. The basic units at each study tree used for insect population and spray deposit sampling were four 15-inch branches taken from mid-crown. Two aluminum plates (6 by 6 inches) and one white Kromekote card were also used to sample spray deposit. Plates and white cards were placed in metal or plastic holders on the top of wire supports and positioned near the ground but above the general level of ground vegetation in the nearest open area adjacent to each sample tree. Additional cards and plates were randomly distributed in the open spaces.

Kromekote cards and aluminum plates have been used as an index of spray coverage on the sample trees and target area by providing data on spray droplet density (droplets/cm^2) and amount of insecticide (gallons per acre, GPA) at ground level. Care must be taken to keep the cards dry because on moist cards the spray spots are not as sharp or distinct and they spread differently. Moist cards tend to warp and, as a result, the reading of spray spots is further confounded (Barry and Markin 1978).

Cards and plates are placed in the field just before spraying. They are prenumbered to correspond with the sample trees, with the number written on the underside. A technique for defining optimum sample size procedures for spray deposit assessment using cards has been developed by Wong (1980). Cards and plates should be left out for at least 45 min after the spray is applied to allow the small spray droplets to settle and dry (Barry and Markin 1978). During the collection, each aluminum plate pair is placed face-to-face (deposit sides together), stored in a slotted box for transportation, and sealed against any light. Kromekote cards can be stored in the same box.

The foliage samples (15-inch or 45-cm twigs) removed from trees for density estimates of pretreatment insect population can be used for the pretreatment foliage assessment samples. One hundred needles should be removed at random from each of the four foliage samples, placed into opaque paper bags marked with the appropriate tree and foliage number, and placed in a larger single paper bag. Foliage samples removed during the first posttreatment density counts, however, cannot be used to assess spray coverage, because they are not taken immediately after spraying and the dye will

have faded. A set of four foliage samples, therefore, should be removed from the sample trees within 3 hours after spraying. These samples should be collected from the four cardinal directions at midcrown and placed in individual paper bags and, with the prespray samples, sent to the laboratory for spray deposit assessment.

Some monitoring of spray drift is recommended. Kromekote cards and aluminum plates can be placed along some of the block edges and in the check areas. In the 1973 Cooperative Tests of Chemical Insecticides for Control of the Douglas-fir Tussock Moth, possible spray drift into the check plots was monitored by placing 10 Kromekote cards and 10 pairs of plates in the 10-tree cluster closest to the nearest treated plots and in a line parallel to the plot border.

Aluminum plates, Kromekote cards, and foliage samples may be processed by the following procedure: spray deposit is washed from each pair of aluminum plates with 10 ml of 95 percent ethanol. The deposit, in terms of concentration (g/ml) of the fluorescent tracers, is measured with a Model 430 Turner Spectrofluorometer and corrected for background fluorescence from prespray samples and percent recovery. Percent recovery is determined by applying known volumes of the formulations to aluminum plates and the deposit then determined by the method described earlier. The ratio of known to measured deposit gives percent recovery.

Concentrations are converted to deposit in GPA, per sampling station. Actual tracer concentrations of a sample of formulation are collected when loading the aircraft. Amount of tracer removed from the plates and area of the plates can be determined with this formula (Maksymiuk 1964):

$$\mathrm{GPA} = \frac{(\mathrm{g.dye} \times 10^{-7})}{\mathrm{ml}} \times \frac{(\mathrm{ml\ wash})\ 94.356 \times 10^{4}\mathrm{ft}^{2}/\mathrm{acre}}{(\mathrm{g.dye/gal})\ (0.25\ \mathrm{ft}^{2}/\mathrm{plate})}$$

Spray drops per square centimeter deposited on the white Kromekote cards can be determined in various ways. Droplets containing fluorescent particles or stains can be counted on the cards. For nonwhite cards, however, a dissecting microscope with an ultraviolet light is needed to illuminate the fluorescent deposit. An image analyzer computer, Quantimet, also may be used for rapid sizing and counting of spray deposit stains on white cards (Waite 1978b). Spray droplets can be sized for atomization analysis. Spray atomization can be determined by the drop-size spectra (D-max) method (Maksymiuk 1964).

An Automatic Spot Counting and Sizing (ASCAS) computer program analyzes the spot count data obtained from spray deposit cards (Young and others 1977). It computes various droplet diameters of the spray cloud in terms of mass media diameter and volume media diameter, spray droplet density, and of volume (ounces and gallons) of spray deposited per acre. The program is on line at the computer center of the University of California at Davis, California, and the U.S. Department of Agriculture's Fort Collins Computer Center in Colorado.

The procedure for assessing spray deposit from foliage samples begins with random selection of 100 needles from each foliage sample collected within 3 hours of treatment. The needles are dried, weighed, and spray deposit removed by washing with 10 ml of 30 percent ethanol. Concentration of fluorescent tracer (g/ml) is measured with a spectrofluorometer. Concentrations are then corrected for background fluorescence, based on the average of the prespray samples and percent recovery. Amount of insecticide on the foliage, in terms of micrograms of insecticide per 100-needle sample, can be estimated if the amount of tracer removed from the needles and the insecticide concentrations from the formulation are known.

MONITORING EFFECTS ON NONTARGET INSECTS

In field and pilot control tests with experimental insecticides it is necessary to monitor effects of chemicals on populations of nontarget organisms for possible registration requirements of the Environmental Protection Agency. Also, it may be necessary to obtain similar data in control programs when registered materials are used at dosages near the upper limit of the material's environmental safety range. Normally, these monitoring studies are the responsibility of various Federal and State wildlife agencies. Also, the Forest Service has assessed the effect of the experimental insecticide on nontarget insects as part of the field tests through cooperators at universities and its own field crews (Shea and others 1982).

These monitoring efforts have used drop cloths to sample arboreal insects and some parasites, malaise traps to sample certain groups of flying insects, and drift nets and surber samples for aquatic insects in small streams. Occasionally, subsamples of the large larval instars are dissected to determine parasitism.

Drop Cloths—Collections of insects from drop boxes or drop cloths placed beneath selected study trees can be examined for the presence of arboreal insects and for parasites and predators associated with the target insect pest. Drop boxes can be particularly valuable in monitoring "knockdown" (insects which are stunned by the insecticide, but recover later) and mortality of arboreal insects.

Malaise Traps—Malaise traps can be constructed and set up in the insecticide treatment plots. Collections made over time from these traps can help determine the qualitative and quantitative changes in insect biomass of those groups normally attracted to such traps. The traps can provide substantial information on the Hymenoptera, Lepidoptera, and Diptera affected by the treatments.

Parasites—A subsample of budworm larvae or other target insect larvae collected during pretreatment and posttreatment

sampling periods can be reared on artificial diet for parasite emergence. Rearing is described in the section on Sampling. Another small sample of larvae from each population sampling should be dissected to determine the incidence of parasitism. Dissecting larvae for parasites is tedious and time-consuming and the procedure cannot be used to process large numbers of larvae as easily as rearing. Dissection is a more accurate procedure in determining parasitism, however, because it can account for all internal parasites, whether or not they emerge. Periodic dissection of small subsamples of larvae can be used to ascertain the accuracy of the larger, more comprehensive rearing program.

The effect of various treatments on internal parasites can be assessed by comparing the incidence of parasitism (expressed as parasites of a particular species per 100 larvae) among the treated populations pretreatment and posttreatment (Williams and others 1969). Incidence of parasitism should also be determined for budworm larvae from the check plots.

Drift Nets—Drift nets can be used to monitor treatment effects on aquatic insects in small streams. Samples should be taken periodically for at least 24 to 48 hours before treatment at various locations along streams running through treatment plots. These samples should indicate the normal diurnal drift pattern and diversity of the aquatic insect fauna in the stream. Periodic posttreatment sampling at the same locations should begin 1 to 2 hours after treatment and continue for at least several days.

Insect populations should be monitored periodically during the season and perhaps several times the following year. If there are ill effects from the insecticides, additional sampling will determine the rate at which nontarget populations recolonize in treatment areas and the rates at which their population sizes increase.

FOLIAGE RETENTION

The major purpose of an operational control project is to minimize injury to the forest resource. One way to assess the effectiveness of the control project is to determine how much defoliation was prevented. How much protection did the treatment provide? Methods to assess foliage retention are generally subjective and, if quantitative, are not very accurate. Despite inaccuracies, however, some assessment of treatment protection should be recorded and analyzed.

Trees and Branches

Currently, foliage retention on trees and branches can be assessed in at least two ways. The first is to visually classify crown defoliation and crown damage on the whole tree (Williams 1967). Crown defoliation and damage are first classified when sample trees are selected in late May before the budworm or target defoliator begins feeding. Crown damage is classified again in August or September during the egg mass sampling period, after the budworm or target defoliator has completed its development. Results are examined to empirically determine variation in time by tree and treatment area. The second method is to compare branch samples. Each new (current year's) shoot on the 15- to 18-inch (45 cm) sample branch collected during each larval sampling period and the current year's shoot on the whole branch collected during the egg mass sampling period is counted and placed in one of the following categories: light damage (0-25 pct defoliation), moderate damage (26-50 pct defoliation), heavy damage (51-75 pct defoliation), severe damage (76-100 pct defoliation) (Williams and others 1971, Grimble and Young 1977). Counts placed into these categories can be tested by covariance analyses to determine significance for each sampling period and area.

Aerial Photography

Although false-color aerial photography is not used to assess defoliation damage in coniferous forests, it has been shown to be a valuable tool for assessing effects of aerial sprays on defoliating pests in hardwood forests. In hardwood forests, false-color aerial photography can provide estimates of foliage saved in spray plots as compared with surrounding untreated areas (Ciesla and others 1971). Aerial photography of each spray plot can be obtained during early June, before defoliation occurs, and again in late July when larvae have completed feeding and defoliation is most conspicuous. Aerial photos should be examined in stereo to compare defoliation in areas protected by spray with that in adjacent forests. Maps showing areas of foliage protection, degrees of defoliation and zones of spray drift in relation to spray plots can be prepared in accordance with procedures described by Ciesla and others (1971).

DATA ANALYSIS AND SUMMARY

Statistical inference and graphic interpretation are relied on to evaluate the effectiveness of the spray applications on target insects, protection of the resources, or both. We recommend those methods described in the section on Experimental Design. We also recommend assessing relationships between insect mortality and various spray deposit parameters such as volume deposits and deposit densities (Hard and others 1978, 1979; Williams and others 1978b; Williams and Robertson 1983; Young 1978). Field effectiveness of various treatments

Table 3—*Significance*[1] *of numbers and mean percent mortality of tussock moth larvae on treated plots of Douglas-fir, British Columbia*

Dimilin treatment (oz a.i./gal)	Days postspray					
	0	7	14	21	28	35
	Mean number of larvae/1000 inches[1]					
2.0	314	165	66	30	7	1
1.0	193	132	106	52	34	15
0.5	276	194	126	65	37	22
Untreated	155	130	86	89	72	60
	Mean percent mortality					
2.0		49.9	79.4	91.4	97.9	99.7
1.0		31.3	46.2	73.9	82.1	92.2
0.5		36.3	61.0	75.8	86.1	89.7
Untreated		18.6	33.2	17.5	15.8	10.0

Source: Hard and others (1978)

[1]Means not connected by a bar are significantly different at the 95 percent level.

Table 4—*Probability of detecting a significant difference in mortality among Dimilin-treated plots, with treatments replicated three times*[1]

Difference in mortality to be detected (pct)		Probability at days postspray				
Range	Deviation from mean	7	14	21	28	35
10.0	5.0, 0, −5.0	< 20	< 20	33	50	89
15.0	7.5, 0, −7.5	21	22	66	83	> 99
20.0	10.0, 0, −10.0	37	40	89	97	> 99
35.0	17.5, 0, −17.5	83	84	> 99	> 99	> 99

Source: Hard and others (1978)

[1]Outlined area shows that experimental power is high if the threshold for adequate power is arbitrarily set at 80 percent of detecting a significant difference.

and control strategies and protection of tree foliage can be evaluated from data on populations, spray deposit, and foliage retention.

Insect Populations

Population data obtained in the sampling procedures described previously can be used in hypothesis testing and also to provide estimates of pre- and postspray population densities and mortality, by tree, cluster, and treatment area for each sampling period. These estimates can be easily obtained with minimum time and expense with a, computer program designed for such analyses (Wilson and Williams 1972). The data format for this program is provided in the *appendix*. The analyses are based on the ratio of insect counts to branch surface area or bud counts. For example, the survival rate (mortality estimate) is given by:

$$r_i = ({}^{X}2i/{}^{Y}2i)/({}^{X}1i/{}^{Y}1i)$$

in which ${}^{X}1i$ and ${}^{X}2i$ denote pre- and postspray insect counts and ${}^{Y}1i$ and ${}^{Y}2i$ denote pre- and postspray measurements of branch surface (or bud counts) for the ith tree.

The budworm population data can be used to

• Compare mortality rates and postspray or residual larval population densities among the treatments over time

• Compare pupal population densities among the treatments

• Compare number of egg masses laid by survivors in treated and untreated populations, among treatments and with those laid by the previous generation.

These comparisons can be tested by ANOVA for each sampling period with budworm population densities and mortalities as the (Y) variables *(tables 3* and *4)*. Multiple comparison techniques can be used to find the means responsible for the differences. If sufficient ranges exist in the prespray population density, analysis of covariance should be used. With prespray budworm population density as the covariate, the effectiveness of the insecticide treatments on a range of population densities can be investigated.

Spray Deposits

Spray deposit data provide the following information for each individual and cluster of trees used in sampling the target insect populations:

• Approximate gallons per acre (GPA) of spray formulation reaching the ground in open spaces beside the study trees

• Densities of the spray droplets reaching the ground in the open beside the study trees

• Volume median or mean diameter (VMD) of the spray droplets reaching the ground in the open beside the study trees

The spray deposit data describe a range of deposited dosage for the study tree sites in each sampling cluster, study plot, and treatment. The insect population data and spray deposit data should be summarized in tabular form *(tables 5* and *6)* (Williams and others 1978b, Young 1978). Field effectiveness of each treatment can be partly evaluated by determining relationships between target insect mortalities (survivorship) and deposit for each insecticide application.

Any possible relationship between deposit and mortality, or residual population levels, or both, should be investigated by regression analysis (Davis and others 1956; Hard and others 1978; Maksymiuk and others 1971, 1975; Young 1978). In all situations, larval mortality or residual population levels are the dependent variable (Y) and the spray deposit is the independent variable (X). For the purpose of such analyses, percent mortality may be converted to probits and deposit values at each sample tree site converted to logarithms *(fig. 19)* (Maksymiuk and others 1975). The graphic relations between the 10-day mortality and spray deposit data *(table 6)* are displayed in *figures 20* and *21* (Young 1978). Additional graphic relations of insect mortality and survival, with dosage rate expressed as ounces of actual insecticide applied per acre

Table 5—*Insect population density and mortality and spray deposit estimates in insecticide tests against the Douglas-fir tussock moth in Washington and Oregon, 1972*[1]

Plot number and location	Treatment dosage	Population density		Mortality	Spray deposit on . . .		
		Prespray	Postspray		Foliage	Aluminum plate[3]	Kromekote card[3]
	Lb/acre	*Insects/1000 inches*[2] *of branch area*		*Percent*	*μg a.i./100 needles*[2]	*Gal/acre*	*Droplets/cm*[2]
				Pyrethrins			
Washington:							
1 (Oroville)[4]	0.05	51.84(22.91)	20.71(40.9)	60.04(35.90)	5.42(9.79)	0.08(.06)	11.08(7.66)
2 (Oroville)	Control	37.10(10.76)	33.28(32.20)	10.30(46.70)	—	—	—
1 (Oroville)[5]	0.05	20.72(12.58)	11.17(19.00)	46.06(18.85)	—	0.06(.05)	—
2 (Oroville)	Control	33.52(9.80)	18.64(22.50)	44.40(23.50)	—	—	—
1 (Oroville)[6]	0.10	51.84(22.91)	11.17(19.00)	78.45(10.31)	—	—	—
2 (Oroville)	Control	37.10(10.76)	18.64(22.50)	49.76(41.43)	—	—	—
				Mexacarbate			
Washington:							
3 (Riverside)	0.15	18.57(61.29)	10.50(13.09)	87.13(17.69)	7.88(4.33)	0.49(.16)	74.43(22.40)
4 (Riverside)	Control	7.60(3.57)	3.67(6.24)	51.71(11.83)	—	—	—
Oregon							
5 (Elk)	0.15	13.49(3.84)	1.98(4.05)	85.29(9.89)	—	0.69(.27)	64.41(25.07)
6 (Ridge)	0.30	45.26(7.25)	17.34(14.48)	61.69(14.40)	—	0.12(.12)	7.71(4.97)
7 (Farmhouse)	0.15	19.22(1.05)	9.27(.77)	51.85(2.84)	—	.32(.16)[7]	29.35(18.34)
8 (Elk)	Control	8.35(1.67)	3.20(2.14)	61.68(10.77)	—	—	—

Source: Williams and others (1978a)

[1] Values in parentheses are standard errors of the mean.

[2] Amount of dye recovered.

[3] Placed in openings in forest canopy nearest study trees.

[4] First application.

[5] Second application. Spray deposits sampled on aluminum plates only.

[6] Two applications compare the prespray population density of the first application with the prespray populations density of the second application. No spray deposits.

[7] Based on treatment of 20 trees only.

Table 6—*Prespray and postspray insects per 100 buds, with corresponding spruce budworm mortality for one spray block with 15 single trees sampled within a block*

Trees	Prespray	Postspray 3 days	Postspray 10 days	Mortality 3 days	Mortality 10 days	Spray deposit	
	Insects/100 buds			*Percent*		*Gal/acre*	*Droplets/cm*[2]
1	6.304	1.301	2.964	0.794	0.530	0.03	7
2	7.087	3.033	2.049	0.572	0.711	0.05	6
3	5.732	1.691	2.239	0.705	0.609	0.06	12
4	5.299	0.702	0.463	0.868	0.913	0.11	12
5	3.759	0.159	0.000	0.958	1.000	0.16	22
6	4.050	0.532	0.986	0.869	0.757	0.05	13
7	5.398	0.426	0.000	0.921	1.000	0.10	21
8	6.558	1.196	0.849	0.818	0.871	0.09	11
9	11.628	3.764	2.711	0.676	0.767	0.07	16
10	11.449	4.745	5.734	0.586	0.499	0.01	4
11	17.886	13.747	8.513	0.231	0.524	0(.01)	1
12	5.702	3.052	3.781	0.465	0.337	0.02	7
13	6.634	1.875	1.760	0.717	0.734	0.03	5
14	5.065	4.351	3.384	0.141	0.332	0.01	4
15	8.244	4.661	5.128	0.435	0.378	0.02	4

Source: Young (1978)

are displayed in *figures 22* and *23*, respectively (Hard and others 1978).

Damage

Amount of defoliation of the new foliage can be classified for each sample branch during each of the larval and pupal sampling periods. These classifications can be examined by covariance analyses to test the amount of defoliation damage resulting from target pest populations within and among the treated and untreated plots. Covariance analyses allow the statistician to handle problems resulting from the statistically unacceptable practice of preselection of check areas and the continued use of Abbott's formula (Abbott 1925).

Check Areas

Check areas are commonly set aside from control projects to estimate natural larval mortality. Spray induced mortality is distinguished from mortality from natural causes by Abbott's formula ($100(x-y)/x$ = % control, in which x = percent survival in check population and y = percent survival in treated population) (Abbott 1925). The result is larval mortality creditable to spraying based on the premise that natural mortality within the two areas occur at similar rates. This is a questionable assumption.

Use of check areas and Abbott's formula are justified in agricultural experiments on annual crops, where environmental conditions between treated and untreated blocks are similar and somewhat more controllable. The check areas, therefore, are representative of the treated areas. But forests present entirely different environmental conditions.

First, representative check areas are difficult to find, particularly in the mountainous West, because of variable topography, local weather conditions, site qualities, and forest stand structures. It is difficult, therefore, to find sufficient similarities between areas so that natural mortalities can be compared with treatment-induced mortalities. Second, workers in control projects tend to spray all areas with high larval populations and, as a result, it is difficult to set such areas aside as checks. Frequently, areas with population densities lower than those to be treated are used as check areas. But this does not provide a comparable situation because natural mortality rates usually correlate with larval densities. Third, if an area with high population densities and environmental conditions similar to areas to be treated is set aside, it is usually so close to the spray blocks that a high probability of contamination by the small, but highly effective, spray droplets exists.

Although there are many difficulties in obtaining comparability among target pest populations in different forest areas, check areas and control populations are necessities in experimental designs to assess treatment effects on target pest populations. Check areas with their resident target pest populations serving as controls are the only areas and pop-

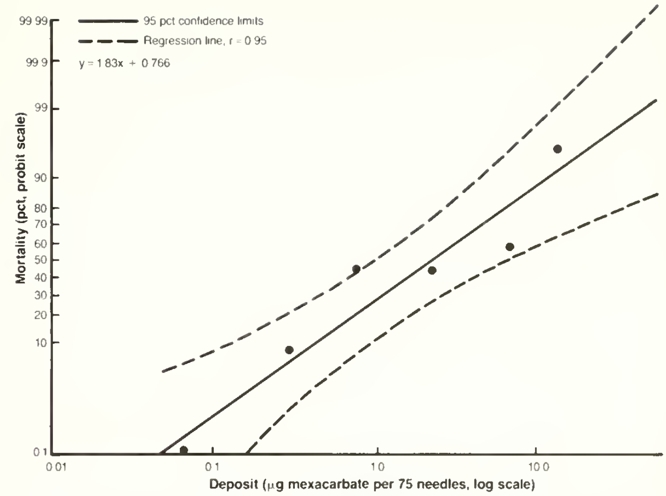


Figure 19—Relationship between average mortality of the spruce budworm in probits and mean deposit in logs for the treated plots. The least squares equation is Y = 1.831X + 0.766, in which Y is the mortality in probits and X is the log (100Xdeposit). The factor of 100 was used to convert the deposit values to a convenient scale for computations (Maksymiuk and others 1975).

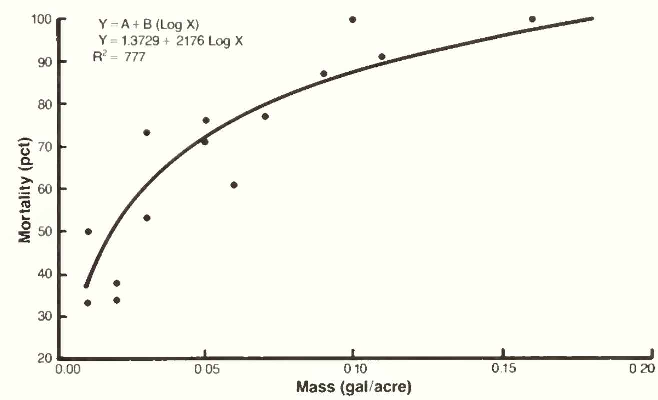


Figure 20—Regression curve of insect mortality over mass recovery in one spray block with 15 single trees (Young 1978).

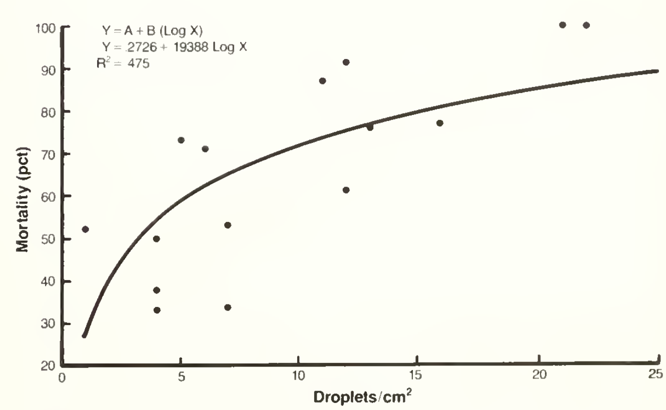


Figure 21—Regression curve of insect mortality over droplets/cm² in one spray block with 15 single trees (Young 1978).

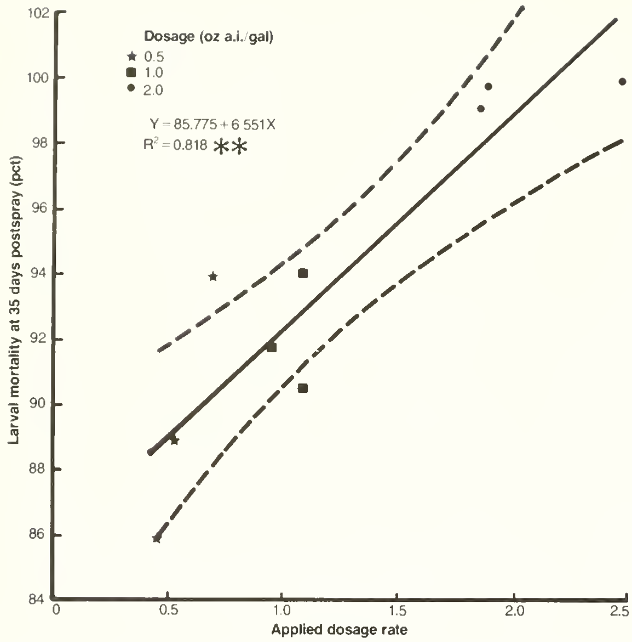


Figure 22—Douglas-fir tussock moth larval mortality, in mean percent per plot at 35 days postspray, is highest in plots treated with 2 ounces of Dimilin per gallon of spray. Applied dosage rate is calculated as a.i. applied per acre = (spray flow in gal) (oz a.i. per gal formulation)/plot area in acres. Dotted lines show 95 percent confidence limits; ** = significant at 1 percent level (Hard and others 1978).

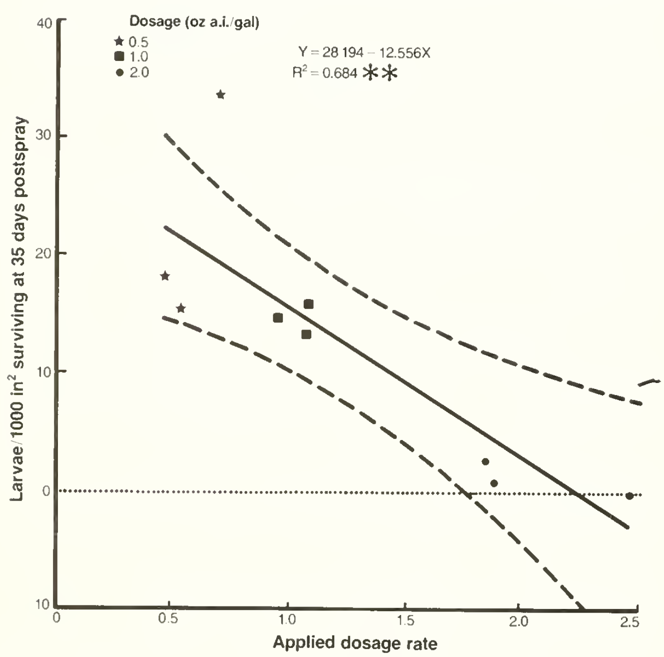


Figure 23—Mean number of surviving Douglas-fir tussock moth larvae/1000 in^2 per plot at 35 days postspray is lowest in plots treated with 2 ounces of Dimilin per gallon of spray. Applied dosage rate is calculated as in figure 22. Dotted lines show 95 percent confidence limits; ** = significant at 1 percent level (Hard and others 1978).

ulations in the experiment without a treatment component. High variability in populations within or between locations in a test lowers the power of the test. The lower the power the more difficult it is to determine differences among treatment effects. The investigator must design the test to increase the power of the test within the limitations of cost. Power can be increased by increasing the number of observations per treatment, by decreasing the number of treatments, and by selecting treatments for comparison with expected large differences in treatment effects. In field tests of insecticides the largest differences are expected to occur between treatment(s) and check or controls (no treatments) (see *Experimental Design*).

Check areas are also worthwhile to determine effects of insecticides on internal parasites of the budworm. Parasitized larvae treated with mexacarbate or DDT survive proportionately much better than unparasitized budworm larvae (MacDonald 1959; Williams and others 1969, 1979). This difference is strikingly illustrated when apparent parasitism of prespray and postspray budworm populations of treated and untreated areas are compared. Check areas are useful to ascertain the apparent effects of insecticides on birds, small mammals, and terrestrial insect populations—particularly those insecticides with high mammalian toxicities.

Sequential Sampling

Sometimes not enough money or workers are available to estimate reliably budworm population densities to determine whether some control action needs to be done. Under these circumstances, sequential sampling may be used profitably. The sequential plan has no fixed sample size and permits classification of pre- and postcontrol population levels within some specified limit of accuracy. Prespray sampling may be done to decide whether control is justified or not. A postcontrol sample is done to determine the effectiveness of a spray project. Of course, the surviving population level is the primary concern in spruce budworm control. Several sequential sampling plans have been developed for the spruce budworm (Cole 1960, McKnight 1970, Morris 1954, Waters 1955). Computer programs are available to develop sequential plans when certain information is available (Talerico and Chapman 1970).

For a sequential plan, sufficient information must be available on:

- The relationship between number of insects per sample unit (15-inch or 45-cm branch) and subsequent defoliation of that unit. This information is needed to set class limits or degrees of infestation for the precontrol sampling.
- Frequency distribution of the prespray insect population (usually negative binomial).
- Frequency distribution of the postspray insect population (a Poisson distribution) (Cole 1960).
- Calculation of the constant *k* by one of several methods (Cole 1960, Mason 1970).
- Determining what constitutes satisfactory or unsatisfactory control, and the class limits.

Table 7—*Sequential table for field use in prespray sampling of spruce budworm larval populations*

Twigs examined	Cumulative number of budworm larvae			
	Class I against Class II		Class II against Class III	
5	5	19	7	48
10	17	32	34	75
15	30	44	62	103
20	42	56	89	130
25	54	68	116	158
30	67	81	144	185
35	79	93	171	212
40	91	105	198	240
45	103	118	226	267
50	116	130	253	294

Source: Cole (1960)

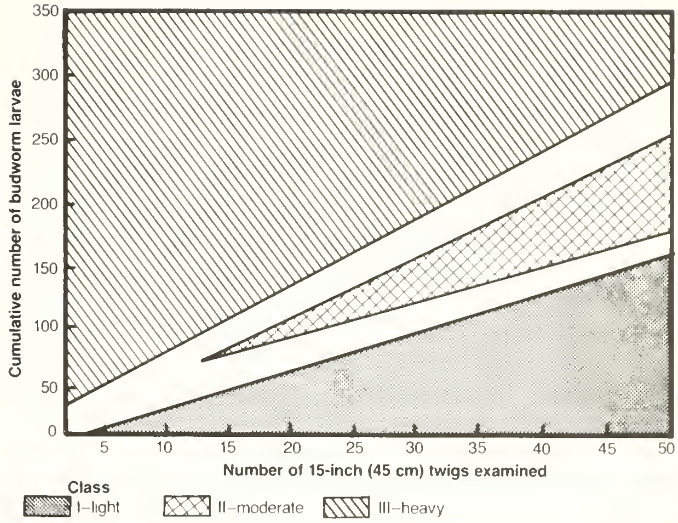


Figure 24—Sequential graph for prespray sampling of spruce budworm populations (Cole 1960).

The sequential plan developed from population-damage studies in southern Idaho (Cole 1960) illustrates the usefulness of this sampling method in budworm control projects *(tables 7, 8, figs. 24, 25)*. A 95 percent reduction of a Class III (heavy) infestation was used as the dividing line between satisfactory and unsatisfactory control. The sequential plan was used in 1965 to classify the mexacarbate and naled insecticide tests in the Bitterroot National Forest, in Montana, as satisfactory or unsatisfactory (Williams and Walton 1968). This National Forest is not far from where the original research was done and, therefore, the plan should apply.

A major problem with sequential sampling in insect surveys is that it assumes that observations are randomly selected. Unfortunately, we seldom know the underlying distributions of our biological subjects, and neither insects nor trees are randomly distributed in the field. When defining a sampling rule for collection of units, therefore, a random order of the sample cannot be ensured. Random order also assumes that all items in the population can be sampled. In fact, it is highly probable that if samples are not selected until one visits the field, all sampling will occur in a limited part of the total area.

All of the sequential plans referred to previously assumed a constant value of the negative bionomial parameter K. Recent studies on the western spruce budworm, however, have shown that K values varied systematically with budworm density for the 4th larval instar, residual pupae, and egg masses (Campbell and Srivastava 1982). New sequential sampling plans have been developed for classifying insect populations with unstable K values, such as the western spruce

Table 8—*Sequential table for field use in postspray sampling of spruce budworm larval populations*

Twigs examined	Cumulative number of budworm larvae: Satisfactory control	against	Unsatisfactory control
15	—		12
20	2		14
25	4		17
30	6		19
35	8		21
40	11		23
45	13		25
50	15		27
55	17		29
60	19		31
65	21		34
70	24		36
75	26		38
80	28		40
85	30		42
90	32		44
95	34		46
100	36		48

Source: Cole (1960)

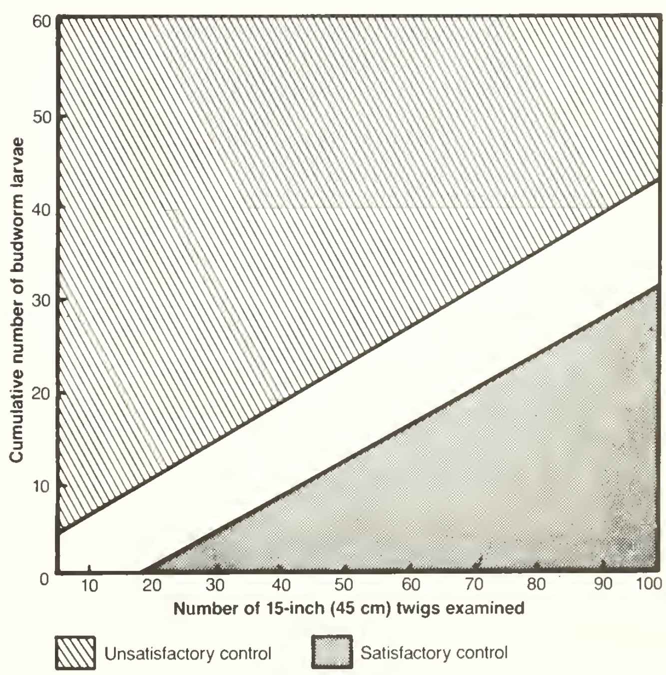


Figure 25—Sequential graph for postspray sampling of spruce budworm populations (Cole 1960).

budworm (Srivastava and Campbell 1982). These sequential sampling plans can classify population levels of 4th instar budworm larvae and egg masses *(tables 9, 10).*

The sequential sampling plan is most reliable in small homogeneous areas (about 5-ha plots) and is used by removing a 45-cm terminal tip from the midcrown of each tree sequentially. A minimum of 1 tree is needed to classify 4th instar larval populations and at least 14 trees are needed to classify egg mass populations.

Srivastava and Campbell (1982) also developed another sampling scheme called "Sequential Count Plans" that can both classify population levels and provide estimates of population density at a prespecified precision level at a lower cost than is needed for fixed-size plot sampling schemes. These sequential count plans provide estimates of budworm densities for 4th instar larvae, egg masses, and residual pupae. For a 20 percent precision level, a minimum of 4 trees must be sampled to obtain larval estimates *(table 11),* a minimum of 9 trees at the pupal stage *(table 12),* and a minimum of 17 trees at the egg mass stage *(table 13).* Density is estimated as the total number of insects at the stopping point per number of trees sampled. This count is converted to density per square meter by dividing the number by 0.082.

Table 9—*Sequential sampling scheme at 95 percent confidence level for larval population (4th instar) on midcrown 45-cm terminal tips*

Trees	Cumulative number of budworm larvae			
	Light	Moderate	Moderate	Heavy[1]
1	2	3	9	10
2	4	6	17	20
3	6	9	25	28
4	9	13	33	37
5	12	16	40	46
10	25	35	77	88
15	39	53	114	130
20	53	72	150	171
25	67	91	187	212
100	285	383	723	825

Source: Srivastava and Campbell (1982)
[1] Infestation limits defined by Carolin and Coulter (1972).

Table 10—*Sequential sampling scheme, at 95 percent confidence level, for egg-mass populations on midcrown 45-cm terminal tips*

Trees	Cumulative number of egg masses			
	Light	Moderate	Moderate	Heavy[1]
10	3	5	9	10
14	4	6	13	15
15	5	7	14	16
20	6	10	19	21
25	8	12	24	26
50	16	24	48	52
60	19	29	58	63
70	23	34	67	74
80	26	39	77	84
90	29	43	87	95
100	32	48	97	105

Source: Srivastava and Campbell (1982)
[1] Infestation limits defined by Carolin and Coulter (1972).

Table 11—*Sequential count plans for estimating 4th instar budworm larvae on 45-cm terminal tips (midcrown)*[1]

Trees (n)	Cumulative number of budworm larvae		
	$D_o = 0.15$[2]	$D_o = 0.20$	$D_o = 0.25$
3	—	—	15
4	—	71	8
7	162	10	5
10	26	8	4
15	16	7	4
20	13	6	4
25	12	6	4
30	11	6	4
50	10	6	4
75	10	6	3

Source: Srivastava and Campbell (1982)
[1] Sampling terminated when cumulative number of larvae $\geq T_n$ at n trees.
[2] Fixed level of precision.

Table 12—*Sequential count plans for estimating residual pupae on 45-cm terminal tips (lower crown)*

Trees (n)	Cumulative number of pupae and pupal exuviae (T_n)		
	$D_o = 0.15$[1]	$D_o = 0.20$	$D_o = 0.25$
6	—	—	30
9	—	45	3
10	—	12	2
15	—	4	2
16	81	4	2
20	12	3	2
25	7	3	2
30	6	3	2
40	5	2	1
50	4	2	1

Source: Srivastava and Campbell (1982)
[1] Fixed level of precision.

Table 13—*Sequential count plans for estimating egg masses on 45-cm terminal tips (midcrown)*

Trees (n)	Cumulative number of egg masses (T_n)		
	$D_o = 0.15$[1]	$D_o = 0.20$	$D_o = 0.25$
11	—	—	47
14	—	—	6
17	—	111	4
20	—	14	3
30	114	5	2
40	14	4	2
50	9	3	2
65	7	3	2

Source: Srivastava and Campbell (1982)
[1] Fixed level of precision.

APPENDIX A—Sample Data Collection Sheet

Area code 1 = Chamberlain; 2 = Belmont; 3 = Check

Data Cards

Field Columns	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	Field description
1–2																	**Card I.D.: 01 or 02** for pre or post-spray
3–4																	**Area code:** see above
5–6																	**Plot-number** 1–3
7–10																	**Tree number**
11–12																	**Tree species** (DF, TF)
13																	**Crown level**
14–16																	**Branch number**
17–20																	**Branch length**
21–24																	**Branch width**
25–28																	**Number of shoots**
29–32																	**Number defoliated**
33–40																	
41–44																	**Total number of WSBW**

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686-178/58210

The Forest Service, U.S. Department of Agriculture, is responsible for Federal leadership in forestry. It carries out this role through four main activities:

- Protection and management of resources on 191 million acres of National Forest System lands.
- Cooperation with State and local governments, forest industries, and private landowners to help protect and manage non-Federal forest and associated range and watershed lands.
- Participation with other agencies in human resource and community assistance programs to improve living conditions in rural areas.
- Research on all aspects of forestry, rangeland management, and forest resources utilization.

The Pacific Southwest Forest and Range Experiment Station

- Represents the research branch of the Forest Service in California, Hawaii, and the western Pacific.

Williams, Carroll B., Jr.; Sharpnack, David A.; Maxwell, Liz; Shea, Patrick J.; McGregor, Mark D. **Guide to testing insecticides on coniferous forest defoliators.** Gen. Tech. Rep. PSW-85. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1985. 38 p.

This report provides a guide to techniques for designing field tests of candidate insecticides, and for carrying out pilot tests and control projects. It describes experimental designs for testing hypotheses, and for sampling trees to estimate insect population densities and percent reduction after treatments. Directions for applying insecticides by aircraft and for evaluating the effectiveness of such applications on target and nontarget insects are explained. The guide can help practitioners improve decisionmaking, efficiency, and safety in the use of insecticides to control forest pests. Although it is based primarily on budworm populations, the information reported is applicable to other coniferous forest defoliators.

Retrieval Terms: insecticide tests, forest defoliators, western spruce budworm, Douglas-fir tussock moth, experimental designs, sampling

United States
Department of
Agriculture

Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-86

Decline of Ohia *(Metrosideros polymorpha)* in Hawaii: a review

Charles S. Hodges   Ken T. Adee   John D. Stein   Hulton B. Wood   Robert D. Doty

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Acknowledgments:

The map of ohia forest decline was drawn at the Hawaii Institute of Geophysics.

Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

January 1986

Decline of Ohia *(Metrosideros polymorpha)* in Hawaii: a review

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IN BRIEF

Hodges, Charles S.; Adee, Ken T.; Stein, John D.; Wood, Hulton B.; Doty, Robert D. **Decline of ohia *(Metrosideros polymorpha)* in Hawaii: a review.** Gen. Tech. Rep. PSW-86. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1986. 22 p.

Retrieval Terms: Metrosideros polymorpha, Plagithmysus bilineatus, Phytophthora cinnamomi, decline, rainforest, Hawaii

Portions of the ohia *(Metrosideros polymorpha)* forests on the windward slopes of Mauna Loa and Mauna Kea on the island of Hawaii were noted to be dying during the late 1960's. Aerial photographic evidence, however, showed some mortality occurred as early as 1954, but most took place between 1954 and 1972. Since 1972 little additional mortality has occurred and the problem is still confined to the same general area. Currently, about 50,000 ha are seriously affected by the problem.

Individual trees affected by decline exhibit several kinds of symptoms, from slow progressive dieback accompanied by chlorosis and reduction in leaf size to rapid death of all or part of the crown. Mortality of fine feeder roots is common on affected trees.

Seven types of decline have been identified on the basis of differential response of the associated rainforest vegetation to collapse of the ohia canopy. Two of the types identified, Bog Formation Dieback and Wetland Dieback, make up more than 80 percent of the decline area. Both dieback types occur in an area of high rainfall and are associated for the most part with poorly drained substrates. However, Bog Formation Dieback develops more slowly, ohia regeneration is poor and mostly of vegetative origin, and treeless bogs are widely scattered throughout the area. Ohia trees in the Wetland Dieback type declined rather suddenly, with the most mortality occurring between 1965 and 1972. Impact on associated vegetation was much less than in the Bog Formation Dieback, and ohia regeneration of seedling origin is good.

Loss of the ohia canopy in some areas has resulted in a decrease in population of some native birds and increased numbers of some introduced birds. Endangered plant species in decline areas probably have also been affected. Introduced plant species have invaded some areas.

Although 90 percent or more of the ohia canopy may be killed, subcanopy species and litter provide ground cover for 95 percent or more of the area. For this reason, decline has had no major effect on watershed values, either in amount of runoff or water quality.

Ohia decline appears to be a typical decline disease in which tree mortality results from a sequence of events that starts with tree stress, which in turn predisposes the trees to attack by organisms that eventually kill them. Poor soil drainage is probably the major cause of stress. The ohia borer *(Plagithmysus bilineatus)* and two fungi, *Phytophthora cinnamomi* and *Armillaria mellea*, attack stressed trees. Low soil nutrient levels, aluminum toxicity, and senescence may also play a role in the overall decline syndrome.

Although ohia decline has severely affected the ohia ecosystem in some areas, an ecosystem dominated by native vegetation probably will continue, but this may differ from that present before decline. Except for control of introduced plants and feral animals that spread these plants, little can be done on a practical basis to ameliorate the effects of decline.

INTRODUCTION

Ohia lehua *(Metrosideros polymorpha)* is the dominant forest tree on all the major Hawaiian Islands, comprising about 62 percent of the total forest area *(fig. 1)*. Although the species is little used commercially, it is invaluable from the standpoint of watershed protection, esthetics, and as the only or major habitat for several species of forest birds, some of which are currently listed as threatened or endangered.

During the late 1960's, large areas of dead and dying ohia were noted along the windward slopes of Mauna Loa and Mauna Kea on the island of Hawaii (Mueller-Dombois and Krajina 1968). The problem, commonly referred to as ohia decline or ohia dieback, was extensive and apparently intensifying (Burgan and Nelson 1972). Since 1970, the decline problem has been studied extensively to determine the extent and cause.

This report summarizes what is known about ohia decline, and discusses the implications for managing declining ohia forests. This report also includes new data on the current distribution of decline on the island of Hawaii.

HISTORY

Although concern for decline in the ohia forest on the island of Hawaii dates only from the late 1960's, groups of ohia trees dying in Hawaiian rainforests were reported as early as 1875 (Clarke 1875). Reports of the Committee on Forestry of the Hawaiian Sugar Planters' Association mentioned widespread mortality of both ohia and koa *(Acacia koa)* along the Hamakua Coast on the slopes of Mauna Kea on Hawaii, and on Maui (Horner 1912, Forbes 1918). The mortality was attributed to "bark beetles."

Large areas of dead and dying ohia were observed on Kauai in the early 1900's (Larsen 1910). Fosberg (1961) mentioned the occurrence of dead and dying trees in Hawaiian forests in certain wet areas on comparatively level or gently sloping ground. One particular stand cited was near Hanalei Valley, Kauai. This is probably one of the same areas on Kauai where Petteys and others (1975) found ohia tree death or crown dieback to be abnormally frequent. Similar declining and dead trees were noted on Kohala Mountain along the Kohala Ditch Trail on the island of Hawaii (Donaghho 1971). Native forests on steep slopes were "holding their own," but on broad, plateau-like ridges between the valleys, many trees were dead or dying. In this area the forest continues to decline *(fig. 2)*.

Davis (1947) observed declining forests in several areas in Hawaii Volcanoes National Park on Hawaii. Photographs in the report showed damage similar to that seen today. Some of the trees appeared to be killed by the ohia borer *(Plagithmysus bilineatus)* but other dead and dying trees were found to be free of insect attack.

All the above reports on ohia mortality contain little or no details of locations, symptomatology, site factors, or other observations to indicate their possible similarity to the current decline on the windward slopes of the island of Hawaii. They do, however, indicate that mortality and decline of ohia on a relatively large scale has occurred in the past in Hawaii.

One historical record of large-scale mortality is well documented and may be relevant to the current dieback problem on the island of Hawaii. In 1906, ohia were dying unaccountably in spots of varying size in the Koolau District on Maui (Territory of Hawaii 1907). By 1907, the affected area covered 1,620 to 2,030 ha and was spreading rapidly. The problem extended from the lower edge of the forest to an elevation of 305 to 915 m between Kailua and Nahiku, Maui (Lyon 1909). Up to 95 pct of the trees in the affected area were dead, including associated subcanopy trees as well as ohia.

The striking feature of the problem was the association of tree mortality with topographic features (Lyon 1909). The affected area consisted of a series of sloping ridges separated by gulches of varying depths. Some of these ridges had broad tops, while others were more or less sharp. Mortality was associated with the broad, flattened ridges, and the forest was still healthy on the sharp ridges, or steep slopes of gulches. Such differences can still be seen today.

Tree death was due to killing of the roots that penetrated more than 5 cm of soil (Lyon 1909). Even trees with apparently healthy tops had roots killed back to near the soil surface. The tissues of the dead roots were deep purple or bluish black. Because no pathogenic fungi or potentially harmful insects were found to be associated with affected trees, such organisms were not believed to be the cause of tree mortality. Rather, the cause was believed to be toxic quantities of hydrogen sulfide and ferrous iron compounds produced in soil by bacterial fermentation under conditions of poor drainage. Lyon (1909) suggested that introduced tree species be planted which might be adapted to the wet sites.

Curran, who visited the area in 1911, concluded that the primary cause of the problem was an exceptionally strong

storm, which struck the area shortly before the mortality problem was noted in 1906 (Curran 1911). He believed that the exposed flat ridges with their adverse soil conditions and excessive moisture, as well as grazing and ditch construction, reduced vigor of trees and made them more susceptible to the storm damage.

Later, Lyon (1918) indicated that ohia mortality may have been due to clogging of soil interstices in the lower strata with fine leached material under conditions of heavy rainfall. Clogging resulted in gradual changes in the substratum which rendered it less suitable for growth. Lyon (1918) again suggested planting such areas with introduced trees that might be more adaptable.

Guernsey (1965) studied the soils in the dieback area and found that an impervious layer was indeed present. This layer was a dominating soil characteristic on slopes less than 25 pct in areas with annual rainfall above 3,750 mm. Depth of the layer beneath the surface was inversely proportional to amount of rainfall. The layer was dense and impervious to air and water movement. He suggested that it is formed over time by reduction of ferric iron to ferrous iron by hydrogen sulfide, which had been produced by anaerobic fermentation.

The Maui forest dieback area still can be easily identified. It consists primarily of 3.5 to 6.0 m tall ohia, apparently mostly of vegetative origin, with various grasses, sedges, and ferns in the understory. The area is boggy with the water table at or near the soil surface. In spite of the severe conditions, the ohia appear relatively healthy, i.e., crowns show little dieback, although some trees appear somewhat chlorotic. This may indicate that there are genotypes within the species that are adapting to the site. Stumps and snags of the original ohia forest, some more than 1 m in diameter, are still evident *(fig. 3)*.

Following Lyon's recommendations, various introduced tree species were planted on a large scale in the area. Two of these, *Eucalyptus robusta* and *Melaleuca quinquenervia*, have survived and grown well, and the soil beneath has dried considerably in comparison with the unplanted area.

Except for general reports of occurrences of ohia mortality mentioned earlier, records of the decline problem on the island of Hawaii were not available before 1954. At that time, aerial photographs were taken of the entire area now known to be affected. These photographs, along with others from the same area taken in 1965 and 1972, allowed evaluation of the severity and rate of decline between 1954 and 1972 on approximately 71,360 ha (Petteys and others 1975). They found that, in 1954, 42 pct of the study area was classified as healthy, while less than 0.2 pct was classified as having severe decline. Most of the areas of moderate and severe declines were concentrated in the northern part of the study area on Mauna Kea. In 1965, 26 pct was classified as healthy and 22 pct as having severe decline, with most areas of severe decline generally in the northeast and southern parts of the study area. By 1972, only 18 pct of the study area was considered healthy, while 48 pct was in severe decline. No evaluation of the occurrence and severity of decline has been published since 1972.

CURRENT STATUS

In 1982, we reevaluated the extent and severity of ohia decline on the island of Hawaii. Three decline classes were subjectively estimated on the basis of percentage of trees dead or showing obvious symptoms of decline: healthy to slight—10 pct or less trees dead or declining, moderate—11 to 50 pct of trees dead or declining, and severe—more than 50 pct of trees dead or declining.

General boundaries of the areas in each decline class were delimited on the basis of two separate overflights in a fixed-wing aircraft and outlined on 1:24,000 orthophotoquads. Later, several overflights in a helicopter more precisely defined the boundaries. Additional information was obtained from interpretation of 1977 color infrared and 1978 black and white aerial photographs and from ground checks. Boundaries were delineated on the orthophotoquads, transferred to tracing paper, and each quad was then photographed, printed at approximately one-fourth of actual size and a mosaic made of the photographs. From the mosaic a map of reduced scale was prepared (see foldout map at the back of this report). Acreages in each delineated area were determined with a planimeter, and verified with an area meter (Lambda Instruments Corp.).[1]

Decline areas on Mauna Loa were easily delimited because of the sharp boundaries between adjacent areas of different decline classes, which generally corresponded to well defined lava flows. On Mauna Kea, boundaries between areas of different decline classes were more diffuse.

The area mapped consisted of eight $7\frac{1}{2}°$ quads and part of two others (see map). This area is larger than that surveyed earlier (Petteys and others 1975), extending slightly farther north, east, and west, but slightly less on the southern end. The earlier survey also included in the forested area historic lava flows—flows for which the dates are known. We did not study these relatively young forests because the ohia trees were considered immature. Some dieback and mortality occur on portions of these young flows, however. As in the earlier study (Petteys and others 1975), former forest areas that had been disturbed by grazing, logging, agriculture or other activities were not considered in the total forest area. Forest plantations also were not considered, but were generally healthy.

Forests in the 1982 study area not influenced by human activity covered 76,915 ha. Of this area, 35.3 pct was considered healthy or had slight decline, 23.7 pct had moderate decline, and 41.0 pct had severe decline *(table 1)*.

The 1982 evaluation cannot be directly compared with the earlier study (Petteys 1975) because of differences in methodology. However, specific photointerpretation points could

[1]Trade names and commercial enterprises or products are mentioned only for information. No endorsement by the U.S. Department of Agriculture is implied.

Table 1—*Status of ohia decline on the island of Hawaii*

Quad	Healthy to slight decline[1]	Moderate decline[2]	Severe decline[3]	Disturbed areas	Historic lava flows	Forest plantations
			Hectares			
Papaaloa	2,136	0	631	1,375	0	51
Keanakolu	4,791	1,962	1,508	913	0	611
Akaka Falls	28	4,766	8,398	566	0	879
Pua Akala	3,085	2,432	2,585	0	0	0
Piihonua	795	5,336	3,527	6,134	1,593	452
Upper Piihonua	4,523	1,157	5,534	0	4,323	0
Puu Makaala	2,290	1,346	5,066	8,762	0	510
Kulani	6,924	791	1,654	2,224	6,391	335
Volcano	1,853	122	2,624	5,861	0	0
Kilauea Crater	728	279	44	3,115	6,576	0
Total	27,153	18,191	31,571	28,950	18,883	2,838

[1]Less than 10 pct of trees dead or showing decline.
[2]11 to 50 pct of trees dead or showing decline.
[3]More than 50 pct of trees dead or showing decline.

be compared with the earlier map because both were of the same scale. These comparisons lead to the following conclusions:

1. The discrepancy in the area of severe decline between 1972 and 1982 is primarily due to differences in interpretation of degree of decline in the ohia-koa forest type. Petteys and others (1975) reported 6,800 ha of severe decline in ohia-koa forest type, but none of this forest type was characterized as having severe decline in the current evaluation. In fact, most of the upper elevation ohia-koa forests (area 5, map)[2] appear to be relatively healthy with only scattered dead trees. The lower elevation ohia-koa forests along the Hamakua Coast (area 6), although characterized as having moderate decline, are considered to be toward the lower end of that classification.

2. Within the area surveyed by Petteys and others (1975), severity of decline apparently has changed little since 1972. The large bog area on Mauna Kea (area 3) was considered to be in severe decline in 1972, with scattered patches of more or less healthy forests. These seem to be deteriorating, however, and there appears to be more open bogs. On Mauna Loa, at least on lava flows covered by thin organic soils (area 8), most of the dieback occurred between 1965 and 1972 and has changed little since then. In the areas most severely affected, up to 90 pct of the trees died and all remaining trees lost most of their crowns.

Dieback continues on Mauna Loa on the lava flows at the upper elevations above the areas of severe decline. The largest areas with active dieback are just below Kulani Prison (area 11), the area shown as moderate dieback in the northeast corner of the Kilauea Crater quad (area 14), and at the upper margin of the area of severe decline on the lava flow just south of the Saddle Road (area 9).

3. Few areas of active decline occur outside the original 1972 survey area, the major exception being the area currently characterized as moderate decline in Keanakolu quad (area 1). This is an area of large ohia with a few koa. Scattered dead and dying trees occur over the entire area.

Few areas of relatively healthy ohia forests remain on the windward slopes of Mauna Kea and Mauna Loa. The largest is in the Puu Makaala area in the Puu Makaala and Kulani quads (area 12). Another area in the Kulani quad lies just south of the 1942 lava flow (area 10). There is, however, some scattered mortality in this area. A relatively large area of healthy open ohia forest with a dense understory of treefern (*Cibotium* spp.) is located in the Upper Olaa Forest Reserve (area 15). Perhaps the healthiest area is a dense forest of rather small diameter trees located just north of the Saddle Road at about 1,370 to 1,675 m elevation (area 7). On the Hamakua Coast, only one extensive area of relatively healthy forest exists. This extends from the southwest corner of the Papaaloa quad into the lower central portion of the Keanakolu quad (area 2). Some scattered mortality is occurring in this area.

Aerial observation of ohia forests on the western and southern slopes of Mauna Loa showed no significant areas of active dieback, although some mortality and dieback of individual trees occur in all areas.

[2]Numbered areas on map represent the general area occupied by the stand condition or type described in the text, not a specific location.

EFFECTS ON ECOSYSTEM

Dieback of Ohia and Associated Plant Species

The effect of decline on ohia can be described in terms of single trees and of the stand as a whole. Symptomatology on individual declining trees varies considerably, even on differ-

ent trees in the same general area. Many trees exhibit a slow general dieback, which begins at the branch tips and slowly progresses downward. This progressive dieback is sometimes accompanied by a progressive chlorosis and decrease in leaf size. Sometimes one or more major branches will die suddenly on trees exhibiting symptoms of progressive dieback or even on trees showing no obvious top symptoms. Another common type of symptom is sudden death of whole trees which previously had few or no evident symptoms. Some trees may produce epicormic shoots along the trunk and major branches after loss of all or most of their foliage. Some of these trees eventually die but others remain alive indefinitely.

Mortality of fine rootlets is common on declining trees, but rootlets greater than 5 mm in diameter seldom die until trees are in an advanced stage of decline. Few major roots die before the entire tree crown is dead.

The chronological and spatial development of decline in ohia forests on the island of Hawaii has been studied in considerable detail (Adee and Wood 1982; Mueller-Dombois 1981; Mueller-Dombois and others 1977, 1980). Seven types of decline were identified on the basis of forest structure, habitat and soil types, and regeneration patterns. Three of these decline types were recognized by both Mueller-Dombois and Adee and Wood, but different names were applied. In the following discussion, names used by Mueller-Dombois are shown without parentheses; those of Adee and Wood are enclosed in parentheses.

These types are discussed here only in general terms. Details of stand structure and lists of plant species encountered can be found in the references just mentioned as well as in publications by Jacobi (1983) and Burton and Mueller-Dombois (1984). Details of physical and chemical properties of the soils associated with declining sites are available elsewhere (Wood 1983).

Bog Formation Dieback (Stunted Ohia Wetland)

Bog Formation Dieback covers a large portion of the east flank of Mauna Kea (area 3), extending from the Wailuku River on the south to above Laupahoehoe on the north between about 610 to 1,460 m in elevation. The area most severely affected is circumscribed by the 7,500 mm isohyet, with the annual rainfall decreasing to about 4,375 mm at the upper limit of declining forests. The general topography of the area is a gentle slope broken by small to large hummocks, ridges, knolls, and cinder cones. Several small to large permanent stream channels cut through the area, but there is little or no gully type erosion.

The characteristic feature of this decline type is the large number of treeless bogs *(fig. 4)*. These bogs vary in size from a few hundred square meters to several hectares. They are larger and more numerous at elevations of 730 to 820 m, which probably represents the zone of highest rainfall. Vegetation consists mainly of *Juncus* and *Carex* with some *Sphagnum*. The introduced grass *Andropogon virginicus* is commonly found on both the treeless bogs and more open surrounding areas.

The ohia in the areas surrounding the bogs is generally unhealthy *(fig. 4)*, but amount of dieback varies considerably with elevation and topography. In the central part of the area at about 730 to 820 m elevation, on the gentle slopes, the ohia is usually of low stature and most trees are dead or in an advanced state of decline. Ohia on ridges and knolls are usually of larger stature, and while mortality and degree of decline are somewhat less than in the surrounding areas, most trees have an unthrifty appearance. The stature of the forest upslope and downslope from the central area increases, and the amount of mortality and degree of decline decreases. However, the forest even at the upper limit of the area is still considered to be in severe decline.

With one exception (area 4), the upper limit of the Bog Formation Dieback area merges abruptly at about 1,460 m elevation with the ohia-koa forest (area 5), which generally can be classified as healthy. Of interest is a large area of moderate declining ohia rainforest (area 4) extending downward from the ohia-koa forest into the severely declining area. The margin between these two areas is not clearly defined.

Dense vigorous mats of gleicheniaceous fern (mostly *Dicranopteris linearis*) occur in large patches in the Bog Formation Dieback area. Most of these appear to be rooted on hummocks or piles of debris. Associated woody vegetation is species poor, and generally unthrifty.

Ohia is maintaining itself in this area by vegetative reproduction from larger fallen trees and by limited seed reproduction on downed trees *(fig. 5)*. More than 50 pct of the young ohia are vegetative in origin. Scarcity of seedlings is probably due in large part to the thick mats of fern. Most young ohia are unthrifty in appearance, and there is some mortality.

Little is known about the soils associated with the Bog Formation Dieback type. Because access is difficult, few detailed soil studies have been attempted. Two soil types were identified in the Bog Formation Dieback type on Mauna Kea (Wood 1983). One was found in the area of highest rainfall on a site covered mainly with *Dicranopteris* spp. and dead and declining ohia. This soil type is characterized by a pan horizon 20 to 30 cm below the surface. The pan is 6 to 12 cm thick, red to grayish-brown, brittle, nonsticky when crushed, and not penetrated by roots *(fig. 6)*. Above the pan are two horizons of muck (undifferentiated whole or decomposed organic matter which is very wet and has little or no structural integrity) which are highly mottled, sticky, gritty, and with a strong hydrogen sulfide smell. The 01 horizon consists of mat-like *Dicranopteris* stems and rhizomes. No abnormal water conditions or mottling were noted below the pan and the horizon is typical of other moderately drained, deep, layered ash soils. The muck horizons were notable for low base saturation, high aluminum, and extremely high iron levels (>3,500 ppm). The pH varied between 4.8 and 5.2. Mueller-Dombois and others (1980) also described a similar hardpan formation at about 30 to 50 cm depth near the edge of the Bog Formation Dieback area in which the soil above the pan was saturated, but below it was drier.

The open bogs on Mauna Kea are characterized by a soil type *(fig. 7)* in which the first horizons are peat, 25 to 60 cm deep, and high in organic carbon, nitrogen, iron, and aluminum. The transition between the peat horizon and the first subsurface horizon is somewhat indefinite. The first subsurface horizon is composed of muck, is slightly darker than the surface, has a hydrogen sulfide smell which increases with depth, and frequently contains large quantities of ohia debris. The peat-muck horizon, which is about 1 m deep, probably sits on a deep, layered ash deposit. The water table persists at or near the surface year-round.

A few scattered wet sites on Mauna Loa were classified by Adee and Wood (1982) as Stunted Ohia Wetland because of the similarity in stand structure to that on Mauna Kea. These sites tended to occupy the lower topographic positions within lava flow complexes. The soils were very poorly drained, rather shallow, and had no restrictive pan horizons but were often underlain by impermeable basalt. At the basalt contact surface was frequently a pronounced gley horizon, indicating poor drainage conditions. The pH of the surface horizon was above 5, and base saturation was very low, but higher than in the bog soils on Mauna Kea.

The chronological development of Bog Formation Dieback is difficult to determine. Photographic records of this area before 1954 are not available. In 1954, a large part of the area now referred to as Bog Formation Dieback was classified as moderate decline (11-40 pct of the canopy dead or dying) with two small areas classified as severe decline (>40 pct dead or dying) (Petteys and others 1975). By 1965, almost the entire area was classified as severe decline.

Current evidence, however, points to a decline process going on for a much longer time. Scattered larger ohia and treefern among the mostly stunted forms now covering the central area are probably relics from a time when the sites were better drained. Numerous dead trees are lying on the soil surface or are partially covered by soil. Perhaps more importantly, considerable ohia debris can be found in the subsurface horizons of the treeless bogs. Although some boggy areas have large numbers of large, dead ohia, which could indicate a relatively rapid transition from closed canopy to bog, most data indicate a more gradual transition over a relatively long period of time. These observations are consistent with the speculation that several bog areas in Hawaii could have developed on level or slightly sloping land in areas of high rainfall after formation of impervious clay layers, above which the soil becomes saturated with water (Fosberg 1961). This process proceeds slowly with succeeding tree generations becoming shorter in stature as the accumulation of water becomes greater, and finally terminates in a low forest, shrub, or sedge bog vegetation. The area of Bog Formation Dieback may eventually evolve into a low stature ohia scrub similar to that described earlier for the Maui dieback area.

Wetland Dieback (Ohia Wetland)

Wetland Dieback is found predominantly on the east flank of Mauna Loa (area 8), between 460 and 1,585 m elevation. The affected area extends from the Wailuku River on the north to just south of the Stainback Highway on the south. Annual rainfall in the area varies from 3,125 to 6,250 mm. Some of the most severely affected areas occur within the lower rainfall limits. The general topography consists of more or less parallel lava flows of various ages. Some of these flows are considered to be "historic," i.e., they occurred within the time frame of recorded history.

The lava flows are generally of two types, pahoehoe and a'a *(fig. 8)*, although integration of the two forms occurs. Pahoehoe flows have a smooth or wrinkled surface and are of more or less the same consistency throughout; however, openings of various diameters and lengths (tubes) occur within the flow. Upon cooling, cracks of various widths and depths form in the surface. A'a flows have a rough, broken surface of lava fragments from a few centimeters to more than a meter in diameter. This broken surface covers a thick, solid interior. A'a flows are generally thicker than pahoehoe flows and the surface topography more uneven.

The consistency of the flows influence greatly the internal drainage through them. On a'a flows, the drainage through the fragmented surface area is rapid. On pahoehoe flows, drainage is limited to the cracks and tubes. On young flows drainage is apparently rapid; however, on older flows, organic material accumulates in the openings and may cause blockage and restrict drainage.

While typical flows of the two types may be readily identified, others may show characteristics of both types. When other factors are added, e.g., ash deposits of various depths, it becomes difficult to draw conclusions about the influence of substrate on drainage and tree growth from casual observations.

A characteristic feature of Wetland Dieback is the relatively rapid dying of high stature, closed ohia forests *(fig. 9)*. Some stands appearing healthy on 1965 photographs were in severe decline by 1972.

Another characteristic feature of Wetland Dieback is its association with the pahoehoe type of lava flow (Mueller-Dombois and Krajina 1968). There are several examples of parallel lava flows of the a'a and pahoehoe types in which the ohia stands on the a'a flows are relatively healthy while those on the pahoehoe flows are in severe decline.

The decline-healthy boundaries between such flows are sharp and apparently static *(fig. 10)*. We observed several cases in which the decline status of the two adjacent flows has remained the same since 1972, and one case in which no change has occurred since 1952, when the ohia stand on the pahoehoe flow was already in severe decline. In all cases we observed, the surface of the a'a flow was 3 to 6 m—and in one case more than 15 m higher than that of the pahoehoe, indicating that the a'a flow occurred later. This is contrary to the flows studied by Jacobi (1983), in which the a'a flow appeared to be partly covered by the pahoehoe. One interesting area involved a portion of an old pahoehoe flow of several hectares in size surrounded by an a'a flow *(fig. 11)*. The ohia stand on the pahoehoe flow was relatively healthy in 1965 but in severe decline in 1972. The surrounding stand on the a'a flow is still healthy.

Not all decline is found on pahoehoe sites, however. Patches of decline may occur on a'a flows (Mueller-Dombois 1980). These appeared to be in small poorly drained areas surrounded by well drained healthy stands.

Because aerial photographs of the wetland dieback are available for only a few years, the exact pattern of symptom progression cannot be determined with certainty. However, a more or less synchronous dieback apparently occurred over large areas, sometimes of several hundred hectares. And there is no evidence of advancing fronts—except possibly at the upper and lower limits of decline—or expanding centers that might be associated with diseases. In one area, what appeared to be a wide ancient pahoehoe flow has been dissected by two later flows (one historic, one prehistoric). The ohia forests on all identifiable segments of the old pahoehoe flow, including isolated kipukas (islands), declined more or less simultaneously *(fig. 9)*.

In the areas of most severe decline, mortality of ohia approached 100 pct over large areas. In other areas, however, mortality ranged from 60 to 90 pct, with the remaining living trees having significant crown loss *(fig. 12)*. In 1976, six transects were established on separate pahoehoe flows that had been in severe decline since 1965 or 1972. All trees that were alive at the time the plots were established were still alive in 1982, and a few appeared to be recovering. Similar observations were reported by Jacobi and others (1983), who found that plots established in 1976 showed no marked increase in decline in 1982.

In contrast to Bog Formation Dieback, the subcanopy vegetation is more diverse in Wetland Dieback, and much healthier. In some areas dense fern mats (mainly *Dicranopteris linearis*) are developing, but these are not so large or numerous as in the Bog Formation Dieback. Some of these mats appear to be declining in patches under developing ohia reproduction. Koa is invading Wetland Dieback areas on well drained microsites and appears to be quite vigorous.

Regeneration of ohia in the Wetland Dieback area is generally very good, and the seedlings and advanced reproduction are healthy *(fig. 13)*. Mueller-Dombois and others (1980) found adequate regeneration (>3,500 seedlings/ha) in all nine releves sampled in this dieback type, with numbers of seedlings per hectare ranging from 3,834 to 26,792. Jacobi and others (1983) reported similar findings. Jacobi (1983) found that 79.4 pct of the subplots measured in one dieback area had ohia saplings 2 to 5 m tall, with a mean cover of 7.6 pct. Ohia reproduction dominates many of the ohia wetland stands but is variable and appears to be limited by dense fern mats and poor drainage on a few sites (Adee and Wood 1982). These factors may prevent the recurrence of a closed ohia forest on such areas, but the general prognosis for the wetland dieback area in this regard is good.

Some introduced plants are invading the Wetland Dieback area. In one study site (dieback and healthy) 20 pct of the plant species recorded were introduced and most of these were recorded only in the dieback stand (Jacobi 1983). However, none of those species encountered were considered a serious threat to native species. Mueller-Dombois and others (1980) found 2 to 12 species of introduced plants in releves sampled in dieback forests. Only *Psidium cattleianum*, which was found mainly at lower elevation sites, was considered to have the capacity to displace ohia.

Wood (1983) recognized two soil types in the Wetland Dieback, both of which were poorly drained. On both soils the surface horizon is generally muck situated in concave pockets of varying surface areas and depths. Both types are usually underlain by pahoehoe lava at various depths, usually at 50 cm or less. The irregular surface topography of the pahoehoe results in considerable variation in soil depth and drainage characteristics of the soils.

One soil has two or more distinct subsurface ash horizons which are high in organic matter, mottled, and have a weak to strong hydrogen sulfide smell. The other soil is always shallower with less ash (if any) in the profile, and has a strong gley horizon at the pahoehoe contact surface *(fig. 14)*. Both soil types are strongly acid, with very low base saturations, very high surface aluminum content, and often extraordinarily high aluminum values (>4,000 ppm) in the lower horizons.

Ohia Displacement Dieback (Ohia-Treefern Forest)

Ohia Displacement Dieback occurs in the Olaa Forest Reserve area on the east flank of Mauna Loa, at about 790 to 1,310 m elevation (area 13). Rainfall in the area varies from 2,500 to 3,750 mm. For the most part the topography consists of gentle slopes and the soils are relatively deep and moderately well drained. The area was previously covered by scattered, large-canopied trees with a dense subcanopy of treefern (*Cibotium* spp.), which still persists.

In 1954, most of the area was considered to be in slight decline, with a few scattered areas in moderate decline (Petteys and others 1975). By 1965, most of the area was in severe decline. Currently, a small strip between about 1,220 and 1,310 m elevation appears to be actively declining (area 14). In contrast to the Wetland Dieback, which occurs on lava flows oriented with the slope, this area of active dieback follows the contours, and is sharply delimited from the healthy ohia-treefern forest above. The name Ohia Displacement Dieback describes a dieback forest in which regeneration of ohia was prevented by the dense treefern subcanopy (Mueller-Dombois 1977). Unlike treeferns on Wetland Dieback and Bog Formation Dieback, the treeferns—as well as other subcanopy species—on this area were unaffected by the factor or factors responsible for decline of the ohia, and form an almost continuous closed canopy *(fig. 15)*.

Adee and Wood (1982) found that germination and establishment of ohia seedlings is not limiting under such conditions. Data of Burton and Mueller-Dombois (1984) also show that a significant number of ohia germinants (86/100 m^2) occur under dense treefern cover, although removal of various amounts of canopy increased the number. Both the above observations agree with Friend's (1980) laboratory study, which suggested that ohia seedlings will survive and grow under even lower radiation levels than those found in the rainforest. However, a closed treefern canopy apparently does constrain the developing ohia population by affecting

Figure 1—Stand of mature healthy ohia, island of Hawaii.

Figure 2—Severely declining stand of ohia on gentle slopes of Kohala Mountain, island of Hawaii. (Photo by Ed Petteys, Hawaii Division of Forestry and Wildlife.)

Figure 3—Short-stature ohia in Maui forest dieback area. Snags of trees that died in the early 1900's can be seen.

Figure 4—Treeless bogs are characteristic of Bog Formation Dieback. Note poor condition of the surrounding forest and heavy cover of *Dicranopteris linearis* in the understory.

Figure 5—Ohia reproduction is mainly vegetative in Bog Formation Dieback.

Figure 6—Water restrictive pan horizons are commonly associated with Bog Formation Dieback. A muck horizon less than 50 cm thick is seen above the hardpan, while below is a moderately drained horizon, typical of deep, layered ash soils.

Figure 8—New lava flows of the pahoehoe (dark) and a'a (light) types differ in physical characteristics.

Figure 7—Soil in open bog areas is characterized by a surface layer of peat 20-30 cm thick, above a muck horizon containing wood debris, which is above a relatively thin sticky gley horizon.

Figure 9—Large pahoehoe flow on Mauna Loa supports massive Wetland Dieback. This flow is bordered on both sides by a younger flow. Small kipukas (islands) showing similar damage are segments of the older larger flow, which were surrounded by the younger flow.

Figure 10—The boundary between declining ohia forest on pahoehoe flow (foreground) and healthy forest on a'a flow (background) is distinct.

survival to the sapling class (Burton and Mueller-Dombois 1984).

Two soil types were encountered in the Ohia Displacement Dieback area (Wood 1983). These deep soils are derived from volcanic ash and are highly leached. In the profiles examined, the lower boundaries were not reached at 1 m depth. Drainage was moderately good, although some indications of pan formation (fractured) were found in the lower horizons. The soils are strongly acid with low base saturation. Aluminum concentration is very high in the surface horizon and even higher (>3,000 ppm) in the lower horizons. A variant soil recognized in one plot has slightly poorer drainage with some mottling and presence of organic pockets in the surface horizons. Iron and aluminum concentrations are very high in the surface horizons.

The long-term outlook for ohia in the Ohia Displacement Dieback type is difficult to assess. Only a severe disturbance to the treefern canopy would result in the ohia size distributions characteristic of a closed ohia forest. Lacking that, some ohia probably will emerge from the more or less persistent bank of seedlings in treefern-dominated stands in openings created by falling ohia snags or treeferns. Young ohia saplings, apparently healthy, can already be seen scattered through the area. Eventually, an open ohia-treefern forest similar to the one above the current dieback forest should develop, but perhaps with fewer ohia.

Dryland Dieback

Dryland Dieback occurs on well drained shallow or deep soil habitats in pocket-like distributions or areas usually less than 0.5 ha in size in Hawaii Volcanoes National Park (Mueller-Dombois and others 1977). This type of dieback appears to be associated with dense stands. Annual rainfall in the area is less than 3,125 mm.

We also recognized a similar type of dieback, which we informally call "hotspots." These also occur in small pockets of 0.1 to 0.3 ha in otherwise healthy dense stands on well drained sites *(fig. 16).* The boundaries between declining and healthy trees are sharp, but with few exceptions, no changes in topography or soil characteristics are discernible at these boundaries. All the trees within the area apparently died at about the same time and the areas have not increased in size in the 12 to 15 years since they were first noted on aerial photographs.

Variable amounts of advanced ohia regeneration occur in many of the Dryland Dieback and hotspot areas (Gerrish and Bridges 1984, Mueller-Dombois and others 1980). Ohia probably will eventually become reestablished on such sites.

Gap Formation Dieback

Gap Formation Dieback occurs on a number of ridges and knolls in the general area of the Bog Formation Dieback (Mueller-Dombois 1981). Such sites are generally elevated above the bogs and usually are covered with relatively tall-stature (over 15 m high) closed or open ohia forests. Formerly vigorous trees usually died in groups and showed no evidence of physical damage. Unlike the Bog Formation Dieback, ohia reproduction is often abundant in the gaps. Mueller-Dombois (1984) suggested Gap Formation Dieback may be equivalent to the Ohia-Koa Dieback type of Adee and Wood (1982).

(Ohia-Koa Dieback)

Ohia-Koa Dieback occurs on the lower slopes of Mauna Kea and extends from the Bog Formation Dieback down to the cane fields (area 6) (Adee and Wood 1982). The transition from Ohia-Koa Dieback to Bog Formation Dieback is rather broad and poorly defined. This is in contrast to the rather sharp transition between the upper limit of the Bog Formation Dieback area and the upper elevation ohia-koa forests, which are relatively healthy. Annual rainfall in the area ranges from 5,000 to 7,500 mm. Drainage varies considerably, but most of the area is considered to be moderately well drained to somewhat poorly drained, with scattered poorly drained areas of variable sizes scattered throughout.

The stands consist of a more or less equal mixture of ohia and koa, with ohia dominating in some areas and koa in others. Dieback in the area is quite variable and affects both ohia and koa. The entire area is currently classified as moderate decline but in reality the area consists of a mosaic of small groups of dead trees interspersed with trees in various vigor classes. Subjectively rating an area of this type is difficult, but it is generally in fairly good health and should be considered on the lower end of the moderate decline scale. Development of decline in the area has been slow, and except for small localized areas, has changed little since 1954.

The subcanopy tree species are typical of other moderately well drained sites and are little affected by loss of the canopy. However, the area has a high population of feral pigs whose rooting activity is seriously affecting the area. Disturbed areas are being invaded by grasses, especially *Setaria palmifolia,* which may limit development of both ohia and koa seedlings. Regeneration is limited in some areas almost entirely to downed trees and hummocks of organic debris. Pig activity may also directly disturb the root systems of both large and small ohia and koa. In the areas just above the cane fields, *Psidium cattleianum* forms extremely dense thickets, which exclude ohia and koa regeneration.

The two soils identified in the Ohia-Koa Dieback are the same as those found in Ohia Displacement Dieback. The soils in the two areas probably were derived from different ash deposits, but no major differences between the two areas were apparent with the limited sampling done.

The prognosis for the Ohia-Koa Dieback area is not clear. Decline in the past has proceeded somewhat slowly and may continue to do so in the future. Not enough information on soil development is available to allow speculation on why impervious hardpans are forming in the Bog Formation Dieback area and not in the Ohia-Koa Dieback area. Formation of impervious pans in the future with the subsequent bog formation in these areas of high rainfall could lead to successional characteristics similar to those of Bog Formation Dieback. A continued high level of pig activity in the area will severely impact ohia and koa regeneration even in the absence

of dieback. Continued encroachment of *P. cattleianum* from downslope will also impact ohia and koa regeneration.

(Pubescent Ohia Dieback)

Pubescent Ohia Dieback occurs on a single lava flow just south of the Saddle Road between 1,220 and 1,370 m elevation (area 9) (Adee and Wood 1982). Rainfall in the area varies from 3,500 to 4,500 mm annually. The lava flow is of the a'a type overlaid by a shallow organic muck, which is apparently well drained throughout the area of dieback.

This particular stand is characterized by a canopy of ohia with pubescent leaves rather than one with glabrous leaves that occurs in other dieback types. Mueller-Dombois and others (1980) suggest that ohia, which is one of the original colonizing species on new lava flows as well as the climax species, occurs as successional races or ecotypes that occupy the roles of pioneer, seral, and late seral dominants, and that Pubescent Ohia Dieback may represent the changeover from one seral type to another. Stemmerman (1983) found that pubescent ohia may represent the colonizing form. The observations of Adee and Wood (1982) support this theory. Trees in the Pubescent Ohia Dieback area are of relatively small stature with a small range of diameters. Floristic remnants of pioneer vegetation of young lava flows, e.g., *Heyotis centrathoides, Dubautia scabra,* and *Styphelia tameiameiae* occur in the subcanopy, indicating that the stand is still going through its primary successional phase. Many of the surviving canopy trees and the majority of the abundant smaller ohia are quite vigorous. If the canopy trees surviving continue to develop normally, they may come to dominate and form a healthy open stand.

Mueller-Dombois (1984) suggested that Pubescent Ohia Dieback may correspond to his Dryland Dieback because of its occurrence on well drained sites. Pubescent Ohia Dieback differs, however, in not being confined to small, more or less circular areas.

Wood (1983) characterized the soil in the Pubescent Ohia Dieback area as having 0.0 to 7.5 cm of organic muck over fragmental a'a lava *(fig. 17).* The surface horizon is strongly acid with low base saturation. Aluminum is high, but generally lower than for other dieback soils. Iron is also high, and calcium levels are among the highest recorded on any of the soils studied. Although the surface horizon is shallow, the effective rooting depth is somewhat greater. Roots may penetrate 0.5 m or more into the fractured a'a lava, which also contains some organic material in the voids between the fragments. No significant differences between physical and chemical soil characteristics could be found between dieback areas and nearby healthy areas.

Watershed Values

In areas of severe decline, 90 pct or more of the ohia canopy may be dead. Such areas are not denuded of vegetation, however. Subcanopy species and litter provide ground cover for perhaps 95 pct of the area (Adee and Wood 1982). Loss of a major portion of the canopy may be expected to reduce evapotranspiration and thereby increase runoff, which in turn might increase sedimentation and reduce water quality. The limited long-term water quantity and quality records for streams draining the area of severe decline, however, preclude determining whether ohia decline has had a significant adverse impact on watershed values.

Limited data on precipitation, streamflow, and sedimentation are available from 1929 to 1980 for the Wailuku River, which drains a portion of the decline area. Before 1954, ohia forests on the Wailuku River watershed showed only slight to moderate decline, but by 1972 they had deteriorated significantly (Petteys and others 1975). Doty (1983) compared annual streamflow relative to precipitation before decline (between 1939 and 1955) and after decline (between 1956 and 1978), floodflow before and after decline, and evaluated current water quality in terms of suspended sediments and certain chemical constituents.

The ratio between precipitation and runoff did not differ before and after decline. Analysis of the precipitation-runoff relationships after storms showed that the percentage of precipitation occurring as runoff was higher for the period before decline than it was after decline. However, even with the lower absolute volume after decline, runoff occurred at a faster rate with a higher peak per unit precipitation. Any changes in flooding could not be related to presence of decline.

No data were available for suspended sediments in downstream water before the advent of forest decline. However, the amount of suspended sediments in the Wailuku River and Honolii Stream, which also drains a decline area, was well within the range of the sediment load of six healthy forested watersheds studied on Oahu (Doty and others 1981). Chemical components of the Wailuku River and Honolii Stream between 1976 and 1979 remained well within acceptable levels. Although the Wailuku River and Honolii Stream watersheds represent only a relatively small part of the affected area on the windward slopes of Mauna Kea, decline of the ohia forests apparently has had little, if any, adverse effect on streamflow and water quality. Reduced evapotranspiration and fog drip resulting from loss of the ohia canopy may have been offset by increased growth of subcanopy species.

Few permanent streams are in the decline area on the slopes of Mauna Loa, and none reach the ocean by surface flow. Although some of the decline sites show evidence of impeded drainage, excess runoff—if any—can rapidly infiltrate the porous substrates downslope from the decline area and reach the ocean through underground channels.

Rare and Endangered Plant Species

Mueller-Dombois and others (1980) recorded several rare and endangered plant species included in the list of Fosberg and Herbst (1975) from various dieback areas. However, their relative abundance was not compared with that in healthy stands. Although no data are available on predecline populations of these species, decreases in the amounts of closed forest habitat, especially within Bog Formation Die-

back and Wetland Dieback, may have resulted in the loss of some taxa.

Populations of Forest Birds

Ohia decline, at least in the large area of Bog Formation Dieback on the east flank of Mauna Kea, significantly affected populations of native and introduced bird species. When compared with adjacent tall closed ohia forests, decline sites had 70 pct fewer 'Apapane *(Himatione sanguinea sanguinea),* 77 pct fewer 'I'iwi *(Vestiaria coccinea),* 47 pct fewer Hawaiian Thrush *(Phaeornis obscurus obscurus),* and 93 pct fewer 'Elepaio *(Chasiempis sandwichensis sandwichensis)* (Scott and others, in press). On the other hand, the populations were larger in decline areas for two introduced species, the Red-billed Leiothrix *(Leiothrix lutea)* (30 pct higher) and the Japanese White-eye *(Zosterops japonicus)* (34 pct higher). Such changes in bird populations undoubtedly were caused by loss of the ohia canopy.

No mention was made of influence on bird populations on the Mauna Loa decline area so presumably no significant differences were noted.

ETIOLOGY

Research on the possible causes of ohia decline has generally followed two lines: (1) decline may be due to attack by insects, pathogenic organisms, or both; and (2) decline may be related to certain ecological and environmental factors within the normal developmental phases of the ecosystem.

Insects and Pathogenic Organisms

Several insects and pathogenic fungi have been investigated as possible causes of ohia decline. Among the insects considered, only the two-lined ohia borer *(Plagithmysus bilineatus)* received significant attention. The ohia psyllids (*Trioza* spp.) and two scolytids *(Xyleborus similis* and *X. saxeseni)* were dismissed early from possible involvement in the decline syndrome (Samuelson and Gressitt 1976).

Several fungi, *Phytophthora cinnamomi, Armillaria mellea, Pythium vexans,* and *Endothia metrosideri* have been suggested as possibly being implicated in ohia decline, but only *P. cinnamomi* has been studied in detail.

Plagithmysus bilineatus

Plagithmysus is a genus of endemic Hawaiian cerambycid wood borers believed to have evolved from a single immigrant ancestor species from the southwest United States or Mexico (Gressitt 1978). Currently, 136 species are recognized. Hosts are known for more than 90 pct of the species, and over 93 pct of these are recorded from a single genus of host plant. Although closely related species occur on different islands and often have the same host association, no one species occurs on more than one island.

Plagithmysus spp. are associated primarily with living plants, and females may lay eggs on the bark of apparently healthy trees. Little is known, however, about the relationship of tree vitality to success of attack.

The two-lined ohia borer *(fig. 18),* is commonly associated with declining trees (Papp and Samuelson 1981). It is 1 of 10 *Plagithmysus* spp. believed to be associated with *Metrosideros* spp. in the Hawaiian Islands, and 1 of 2 on the island of Hawaii. Its only host is *M. polymorpha,* but the insect is widely distributed on the island (Stein 1983) *(fig. 19). Plagithmysus abnormis* was also reported from *Metrosideros* on Hawaii (Gressitt and Davis 1969), but its biology is unknown. It was not encountered during studies on ohia decline.

Papp and others (1979) studied the association of *P. bilineatus* and the root fungus *Phytophthora cinnamomi* with ohia trees on 100-tree transects on several sites, both within and outside the area of intensive decline. The roots of 40 trees on each transect were examined to determine the degree of necrosis, and sections were cultured for the presence of *P. cinnamomi.*

Larvae of *P. bilineatus* were found on declining trees at each site. The percentage of trees infested as well as frequency of attack was greater on trees with the poorest crown conditions. However, 14 pct of the apparently healthy trees were attacked, and 25 pct of severely declining trees were not attacked by *P. bilineatus.*

Most galleries of *P. bilineatus* in standing trees were abortive, containing only dead larvae or pupae (Papp and Samuelson 1981). Percentages of galleries with living larvae or pupae in standing trees varied from 4.3 to 30.0 pct and averaged 11.4 pct. In slash of previously felled trees, on the other hand, 89 pct of galleries contained living larvae or pupae. Successful attack (those in which larvae pupated) was much higher in severely declining trees (31.1-35.3 pct) than in apparently healthy trees (2.0-4.8 pct).

The results of the transect study indicate that *P. bilineatus* apparently prefers declining trees as oviposition sites. Other studies support this conclusion. Papp and Samuelson (1981) found that *P. bilineatus* adults are rapidly attracted to felled trees, sometimes within a few hours of felling, even in healthy forests. Nagata and Stein (1982) used guy wires to brace several healthy trees in an upright position before severing them at the base. The saw kerf was sealed and all traces of sawdust were removed. The severed trees as well as adjacent uncut trees were sprayed with tanglefoot adhesive to trap any adult beetles attracted to the trees. Beetles were attracted to all cut trees before the trees showed any visual symptoms of stress. Beetles were not attracted to adjacent control trees. Survival of larvae increased from 10 pct in healthy trees to 83 pct in severed trees (Stein and Nagata 1985). The attraction of beetles to severed trees apparently was a specific response to chemical stimuli produced by physiological changes taking

place after severing. Similar changes presumably occur in declining trees as well. Beetle preference for oviposition on stressed trees may provide a mechanism whereby larvae have a greater chance of survival.

Plagithmysus bilineatus is probably not the primary incitant of ohia decline. This insect may, however, hasten the death of stressed trees or cause the death of stressed trees that otherwise might have recovered. Galleries of beetle larvae are confined to the cambium-phloem tissue or to the outermost sapwood. These galleries typically spiral around the tree trunk or branches in barber-pole fashion and commonly reach lengths of 2 m or more *(fig. 20).* Such galleries effectively girdle the stem or branch, which may result in the death of some trees or portions of trees. The number of declining trees that might have survived in the absence of attack by the ohia borer has not been determined, however.

The ecological niche of *P. bilineatus* in the ohia forests on the island of Hawaii includes the spectrum from apparently healthy trees to living trees under some kind of stress (Papp and others 1979). In forests without decline, the insect attacks apparently healthy, overmature, or suppressed trees, or individual trees weakened by adverse localized conditions. However, a preferential attraction to stressed host material is evidenced by the rapid appearance of adult beetles on cut trees in healthy forests (Nagata and Stein 1982, Papp and Samuelson 1981). The same behavior may be shown in the initial or latent phases of ohia decline. As decline intensifies, attacks become more numerous in the progressively weakened trees. Where trees decline somewhat rapidly and synchronously over an extensive area, the available opportunet substrates for oviposition may exceed the resident female population, and a certain proportion of declining trees will escape attack.

Phytophthora cinnamomi

The fungus *P. cinnamomi* is widely distributed and reportedly is pathogenic to a large and diverse variety of plant species (Zentmeyer 1980). Of particular relevance to ohia decline is the implication of *P. cinnamomi* in several dieback and decline type diseases, including these: littleleaf disease of shortleaf pine in the southeastern United States (Zak 1961), jarrah dieback in western Australia (Podger 1972), dieback of *Eucalyptus* spp. in eastern Australia (Weste and Taylor 1971), and rainforest dieback in Queensland, Australia (Brown 1976).

Phytophthora cinnamomi was first reported from Hawaii in pineapple fields on Oahu (Sideros and Paxton 1930). The fungus subsequently was found on a wide variety of hosts on the other major Hawaiian Islands. Before its possible implication in ohia decline, only Mehrlich (1936) had reported it in the forests of Hawaii.

Kliejunas and Ko (1973) were the first to recover *P. cinnamomi* from roots of ohia. The fungus was isolated from necrotic rootlets from declining ohia on wet sites but not from healthy or declining ohia on dry sites. Bega (1974) also recovered *P. cinnamomi* from roots of declining ohia as well as those of several associated species. Kliejunas and Ko (1976a) found the fungus in roots of declining trees in 32 of 35 decline areas sampled and occasionally from healthy trees in declining and healthy forests. It was also recovered from soil in 72 of the 75 decline sites sampled and from 13 of 15 sites on which trees were apparently healthy.

Further studies (Kliejunas and others 1977) showed *P. cinnamomi* to be widely distributed in the native forests on the other major Hawaiian Islands *(fig. 21).* The presence of the fungus was correlated with wet soils but not with health of the forest canopy, or subcanopy density or composition. It was isolated from 97 pct of the poorly drained sites on all the islands, regardless of the presence of decline, but not from excessively drained sites. The fungus was commonly recovered on Maui (Kliejunas and others 1977) from the area where severe decline occurred in the early 1900's (Lyon 1909). Currently, this area supports a sparse, low stature ohia forest in a relatively good state of health.

In areas where sharp boundaries exist between adjacent declining and healthy forests, such as those described in the section on Wetland Dieback *(figs. 8 and 9),* little difference can be found in populations of *P. cinnamomi* between healthy and declining sites (Hwang and Ko 1978b, Kliejunas and Ko 1976a).

Papp and others (1979) analyzed trees on each of eight sites with varying degrees of decline for rootlet mortality and the presence of *P. cinnamomi.* They found that *P. cinnamomi* was present on 80 pct or more of the trees on the two sites with the greatest decline and the highest percentage of rootlet mortality. These two sites were also the most poorly drained. On the sites with less decline, rootlet mortality was less but was not related to presence or absence of *P. cinnamomi.*

In addition to ohia, *P. cinnamomi* has been isolated from roots of a large number of indigenous, endemic, and introduced species associated with ohia (Bega 1974, Kliejunas and Ko 1976b). The health of these associated species was not detailed, although some species had decline symptoms (Kliejunas and Ko 1976b).

Seedlings and small plants of some species are susceptible to *P. cinnamomi* when artificially inoculated. Kliejunas (1979) compared the relative susceptibility to *P. cinnamomi* of ohia and several other endemic and introduced species. In this study ohia was judged to be moderately susceptible to the fungus while the other endemic species tested were highly to moderately tolerant. *Eucalyptus sieberi, E. baxteri,* and *E. marginata,* reported to be highly susceptible in Australia (Podger 1972, Weste and Taylor 1971) were also susceptible in Hawaii.

Current knowledge on the relationship of *P. cinnamomi* to ohia decline can be summarized as follows:

- The fungus has wide distribution in the native rainforests on all the major Hawaiian Islands. Although it has been recorded in Hawaii only since 1925, it probably has been present for much longer. The fungus may be indigenous to the islands because of the relative tolerance of most of the endemic plant populations (Kliejunas 1979). It is found in remote roadless areas but could have easily been spread there by the activity of feral pigs, which are numerous in the

Hawaiian rainforests. The fungus has been recovered from the feet of feral pigs (Kliejunas and Ko 1976b). Surface movement of water during heavy rains also could result in rapid spread.

• Although *P. cinnamomi* is commonly associated with declining ohia on the island of Hawaii, it is also present in apparently healthy ohia forests on Hawaii and other islands. Thus, there is no constant association between the presence of *P. cinnamomi* and ohia decline. Rootlet mortality on declining ohia is usually high and *P. cinnamomi* can be isolated from dead and dying rootlets. However, rootlet mortality could also result directly from poor drainage and subsequent low soil aeration characteristic of most decline sites, as well as from other factors. *Phytophthora cinnamomi* can colonize pieces of ohia shoot tissue (2 mm in diameter) placed in infested soil, so rootlets killed as a result of poor soil aeration or other factors possibly could later be colonized by the fungus (Hwang and Ko 1978a). The fungus was also isolated after 1 year from 50 pct of the artificially inoculated ohia root pieces buried in the soil, demonstrating its saprophytic competitiveness.

• *Phytophthora cinnamomi* has been isolated from a wide variety of native and introduced plant species in Hawaii. In inoculation studies that used seedlings or small plants, ohia was determined to be "moderately susceptible" to the fungus when levels of inoculum were high. Seedlings grown in soils from decline areas with lower natural levels of inoculum had less root necrosis and no significant growth loss. Associated native plants were found to be even less susceptible. Large numbers of ohia seedlings and saplings are present on many sites where mortality of the ohia overstory was 75 pct or more and where *P. cinnamomi* was present in relatively high populations. These seedlings and saplings are growing vigorously and display no crown symptoms.

The exact role of *P. cinnamomi* in the ohia decline syndrome is difficult to determine. It appears unlikely, however, to be a primary factor in initiation of the problem. The fungus probably plays a secondary role, attacking rootlets of trees already under stress, and contributes to a greater or lesser degree to the overall decline sequence.

Armillaria mellea

The root fungus *Armillaria mellea* is distributed worldwide in temperate and tropical regions and attacks a wide range of hosts. The fungus has been known to occur in Hawaii since 1963 (Rabbe and Trujillo 1963), when it was reported to have attacked several *Pinus* spp. on land cleared of native forest. The fungus later was reported to occur on ohia and other native and introduced tree species (Burgan and Nelson 1972, Laemmlen and Bega 1974).

We have observed *A. mellea* to be widely distributed throughout the decline area in areas above approximately 1,200 m elevation, on both dead ohia trees and those exhibiting symptoms of decline. However, it was not consistently associated with declining trees. In some areas, as many as 80 pct of dead and declining trees had typical mycelial mats of the fungus, while in nearby areas severely affected by decline, the fungus could not be found. These observations indicate that *A. mellea* cannot be considered a primary cause of decline, but almost certainly contributes to mortality of declining trees in areas where it is present. A similar role for *A. mellea* has been proposed in several dieback and decline diseases in the Eastern United States (Houston 1973).

Pythium vexans

Pythium vexans is widely distributed on the island of Hawaii, and has been isolated from roots of declining ohia trees (Kliejunas and Ko 1975). The fungus was pathogenic to ohia in greenhouse inoculation studies, causing root necrosis or seedling death or both. However, the fungus also occurs in areas supporting healthy ohia forests, and could not be recovered from all areas exhibiting severe decline. *Pythium vexans* probably plays no major role in ohia decline, but some minor root necrosis and contribution to the overall decline syndrome cannot be discounted.

Endothia metrosideri

Roane and Fosberg (1983) described a new fungus, *Diaporthopsis metrosideri*—later changed to *Endothia metrosideri* (Barr 1983)—from recently dead ohia trees in Hawaii Volcanoes National Park. Later, Fosberg (1983) speculated that the fungus was pathogenic to ohia and suggested that it posed a serious threat to this species. However, no inoculation studies to confirm pathogenicity have been carried out thus far. Even if *D. metrosideri* is pathogenic, any connection with ohia decline is doubtful. Fruiting bodies of the fungus are fairly conspicuous, and we have not observed them on dead or declining trees in the major decline areas.

Climatic, Edaphic, and Environmental Factors

Rainfall

Petteys and others (1975), in their initial survey for ohia decline, found a correlation between rainfall and ohia decline index in each of 3 years (1954, 1965, and 1972) for which estimates were made. Decline index, on the basis of canopy loss, increased as precipitation increased. A similar correlation was found between increase in elevation and increase in decline index. This correlation is consistent with the occurrence of generally higher rainfall at the midelevations, where decline is most severe; however, regressions of decline index on precipitation and elevation for each of the 3 years showed that they explained only 4 to 9 pct of the variation. Precipitation and elevation are probably not the limiting factors in the decline problem.

Doty (1982) studied the precipitation records from the southeastern and windward sides of the island of Hawaii between 1890 and 1977. Although a consistent long-term downward trend in precipitation was noted for the windward side of the island where ohia is declining, relationships between patterns of precipitation and occurrence of decline were not significant.

Analysis of climatic data from the ohia decline area by Evenson (1983) for the period 1891-1982 showed a large fluctuation in relative water availability, compared with that during the median year. Such fluctuations could produce either flooding or drought conditions across the decline area, depending on soil conditions. While the unusually wet years in the middle to late 1950's and the dry years 1958-59 and 1962 could have contributed to development of Wetland Dieback and Dryland Dieback, respectively, Evenson concluded that much more information is necessary before the exact role of climate can be determined.

Nutrition and Toxicity

Kliejunas and Ko (1974) applied various combinations of fertilizer treatments to single trees in intermediate or advanced stages of decline and to plots containing trees in various stages of decline. Treatment of individual trees with complete fertilizer resulted in production of numerous new leaf buds after 6 weeks, and new vigorous healthy leaves after 3 months. Foliar applications of nitrogen, phosphorus, and potassium (NPK) plus a micronutrient solution also resulted in new buds and leaves, but fewer than resulted from ground application. The new growth produced in response to the fertilizer applications remained vigorous after 1 year.

In a separate study, declining trees responded to both complete fertilizer and NPK with micronutrients, but not to N, P, K, or micronutrients alone (Kliejunas and Ko 1974). Trees showed some response to NP but not to NK or PK.

In plot studies with small trees, application of complete fertilizer resulted in development of numerous new buds on trees in all stages of decline, and fertilized trees were still producing new buds 11 months later. When larger trees were fertilized, only a few new buds were visible after 3 months. However, 4 months after a second application, a definite response was evident.

When fertilizer was applied in combination with various nematicides and fungicides (Nemagon, Difolitan, Benlate, or Dexon)[3] (Kliejunas and Ko 1976a), response of declining trees to fertilizer alone was similar to that in a previous study (Kliejunas and Ko 1974); but no response was obtained with any of the nematicides or fungicides. However, fertilizer plus fungicides resulted in greater response than did fertilizer alone.

Most of the trees used in the above-mentioned experiments were small (less than 2.5 m high) and were growing on relatively young historic lava flows. Whether the response of these young trees to fertilization can be extrapolated to trees in high stature ohia forests suffering severe decline is uncertain at best.

Gerrish and Bridges (1984) established in 1979 a combination thinning-fertilization treatment in three closed mature ohia forests on well drained sites (two on pahoehoe substrates and one on eutrophic ash). At each site, trees with partial foliage loss (10-60 pct) were selected for fertilization or release from competitors or both. Fertilized trees received NPK fertilizer of 16-16-16 composition at the rate of 784 kg/ha in 1979 and three additional applications of 392 kg/ha of Keaau 19 (approximately 12-27-7 plus micronutrients) in 1981 and 1982. The mortality rate of treated trees 2½ years after the first application of fertilizer was no lower than that of the untreated trees. However, application of fertilizer at two sites and fertilizer plus thinning at the third site significantly increased mean annual diameter growth. These increases, however, were small—less than 2 mm over the controls.

Little information is available on the nutrient status of soils in the ohia decline area. Wood (1983) did not find any consistent differences between declining and healthy sites. In general, the levels of all nutrients were low in all soils examined. Nitrogen levels varied from 1.2 to 2.1 pct in the surface horizons but generally were less than 0.5 pct in lower horizons. Phosphorus varied from 3 to 28 ppm in the surface horizons but was seldom more than 8 ppm in the lower horizons. Potassium levels ranged from 64 to 488 ppm in the surface horizons but were generally less than 200 ppm. Levels in lower horizons were usually less than 70 ppm. Levels of calcium (284-3,011 ppm) and magnesium (67-497 ppm) were variable in the surface horizons. Manganese levels were generally low (6-30 ppm) but occasionally were high (78-232 ppm). Aluminum—extracted at pH 4.8—also varied, with the surface horizons containing from 272 to 1,578 ppm. Unlike other elements, however, aluminum levels increased dramatically in the lower horizons, sometimes reaching levels of more than 4,000 ppm.

All of the soils studied had relatively high cation exchange capacities, but low base saturation percentages. Organic carbon contents in surface horizons were high (16-68 pct), suggesting limited microbial activity on most sites. Most of the soils had very low bulk densities (0.1-0.2 g/cm^3) and high water-holding capacities (80-90 pct by volume), regardless of the parent material (ash, basalt, or organic). Upon drying, soil shrinkage can be more than 70 pct of its original volume.

Foliar nutrient levels in healthy ohia differed little among sites and soil types (Mueller-Dombois 1981, Wood 1983). Levels generally were 5,600 to 7,500 ppm for nitrogen, 800 to 1,200 ppm for phosphorus, 4,500 to 7,500 ppm for potassium, 3,500 to 3,800 ppm for calcium, 1,200 to 1,500 ppm for magnesium, and 19 to 23 ppm for aluminum. Differences in foliar nutrient levels between healthy and declining trees were not consistent, although aluminum levels from selected declining and dead trees were up to three times higher than those in healthy trees. Preliminary data of Mueller-Dombois (1981) indicate that concentrations of most elements tend to decrease in leaves of declining trees but that aluminum and manganese increase.

Wood (1983) found that aluminum concentrations in ohia roots were up to 20 times greater than levels in leaf tissue. Smaller roots ($<$5 mm in diameter) contained greater amounts of aluminum than did larger 5- to 10-mm roots. Roots of one declining tree had an aluminum concentration of 977 ppm, but concentrations in some healthy trees

[3]This report neither recommends the pesticide uses reported, nor implies that they have been registered by the appropriate governmental agencies.

exceeded 700 ppm, and no consistent pattern between aluminum concentration and health of tree was evident.

The high levels of aluminum found in the roots of ohia trees could affect the uptake of other nutrients by the tree and further aggravate the poor nutritional status of the tree due to low levels of nutrients in the soil. The levels encountered also could be potentially toxic to fine feeder roots. Death and blackening of such roots commonly associated with declining trees are similar to symptoms associated with aluminum toxicity in other plants (Foy 1971).

More data on both soil and foliar nutrient levels are needed before any definitive conclusions can be reached concerning the relationship between nutrition and ohia decline. Even though both soil and foliar nutrient levels are low in most cases, they are not low enough to have caused the massive death and decline of the ohia forest. Except possibly for certain metals like aluminum, a sudden change in nutrient status that would account for the relatively rapid decline is not likely. However, the generally low nutrient status of the ohia forests possibly could make the trees more vulnerable to other stresses and thus play a secondary role in the decline syndrome.

Soil Drainage

The association of some types of ohia decline with poor soil drainage is well documented. Bog Formation Dieback and Wetland Dieback, which make up more than 80 pct of the area in severe decline (map, *table 1*), are both characterized as being on poorly to very poorly drained sites (Adee and Wood 1982, Mueller-Dombois and others 1980). These decline types also generally occur in areas of the highest rainfall on the island, although portions of the Wetland Dieback occur on more mesic sites. Most of the mortality in the Ohia-Koa and Dieback Gap Formation Dieback types is also associated with poorly drained sites. Ohia Displacement Dieback and Pubescent Ohia Dieback, which also contain areas classified as being in severe decline, occur on moderately to well drained substrates. However, these two types make up only a small percentage of the total area in decline.

The drainage characteristics of the various soils that support healthy and declining ohia forests—for the most part—have been subjectively estimated on the basis of such visual parameters as soil color, structure, vegetation, and topographic position. Doty (1981), however, established wells consisting of 2.5-cm diameter tubing to study the relationship of groundwater levels to ohia decline. The wells were installed at seven locations, each of which contained two to nine individual wells. These locations represented a wide range of soil conditions and vegetation, and included both healthy and declining ohia forests. Ground water levels in the wells were observed weekly, and the weekly high level was determined by the adherence of finely ground cork on a wooden dowel placed in the well.

One year after installation of the wells, water levels in the individual wells fluctuated considerably, sometimes even among wells in the same location. In general, however, the organic muck soils that supported severely declining trees were saturated to within 5 to 10 cm of the surface more than 50 pct of the time. Some wells had water levels at this depth more than 90 pct of the time. On healthy sites over pahoehoe lava, water levels were lower than 22 cm 90 pct of the time. On a'a lava sites, the water level was never less than 36 cm below the soil surface, regardless of whether the site supported healthy or declining forests.

Wood (1983) continued to monitor the wells installed by Doty for an additional 1½ years. These new data generally confirmed the high variability previously reported by Doty (1981). The wells were excavated at the end of 2½ years to enable better characterization of the microsite conditions at each well. Water levels were charted for four wells on soils that were representative of those on which the major types of decline are found *(fig. 22)*.

Water levels on the very poorly drained bog soils, where all the Bog Formation Dieback sites are located, were at or above the surface 90 pct of the time *(fig. 22A)*. Little difference was apparent between the highest and average weekly water levels.

The mean water level typical of Wetland Dieback was only 7.6 cm below the soil surface and above-surface water levels, which occurred during storm periods, were common *(fig. 22B)*.

On the deep ash, moderately well drained soils typical of the Ohia-Koa and Ohia Displacement Diebacks, ground water levels rarely stayed near the soil surface for long periods of time. An example from one of the shallower ash soils *(fig. 22C)* shows that ground water levels remained more than 30 cm below the soil surface 90 pct of the time.

Most of the healthy sites sampled were on better drained sites; however, both Pubescent Dieback and Dryland Dieback also occur on well drained soils. The ground water levels in these well drained soils were usually well below the soil surface *(fig. 22D)*, although the highest water level attained during any weekly period occasionally reached the surface or exceeded it. Even during storm events, these periods of high ground water periods were brief, never lasting beyond the duration of the storm.

Within the scope of the study established by Doty (1981), no healthy ohia forests were sampled on the more poorly drained sites, and certain dieback types were associated only with continuously high ground water levels. While the exact role of drainage or high water levels has not yet been clarified, the poor drainage conditions associated with declining ohia on most areas could severely affect root functions and subsequent tree vigor and health.

Synchronous Cohort Senescence Theory

Early in the course of research on ohia decline, Mueller-Dombois (1974, p. 10) suggested that pathogens were not the primary cause of the problem, even though they may be involved as secondary agents that operate after tree vigor is reduced by some other cause. Instead, he proposed ". . . that the ohia dieback is a normal phenomenon, a developmental stage in primary succession of an isolated rainforest ecosystem." This hypothesis developed in succeeding years (Mueller-

Dombois 1980, 1981; Mueller-Dombois and others 1977, 1980) and recently culminated (Mueller-Dombois 1982, 1983, 1984; Mueller-Dombois and others 1983) in a theory termed "synchronous cohort senescence." In this theory, a generally even-age or even-stature stand of ohia (cohort) results from a catastropic disturbance such as a lava flow, ash deposit, or hurricane. Seed for regeneration comes from adjacent stands. The stands develop and eventually reach maturity, after which senescence begins. Different forms or manifestations and lengths of senescing periods may be programmed into the life of certain species or may be manifested differently in the same species when it grows on different habitats and under different environmental stresses (Mueller-Dombois 1984). Inherent in this concept is the possibility that environmental factors may affect the initiation of senescence (Mueller-Dombois 1983), which in some cases may be reversible (Mueller-Dombois and others 1983).

For synchronous dieback of the ohia cohort to occur, a second disturbance is necessary after the onset of senescence. This disturbance could be a fluctuating site factor such as a storm, temporary flooding, or soil drought, which might not seriously affect a vigorously growing stand but might trigger decline in a senescing stand. Such a disturbance may also act as an additional synchronizing factor. After decline is triggered, secondary pathogens or insects may attack the weakened trees. The severity of the second disturbance and of the attack (if any) of secondary organisms may influence the amount of damage the stand (canopy) sustains, the speed and pattern of tree mortality, and whether there is partial recovery.

In summary, the dieback mechanism proposed for ohia decline involves (1) cohort senescence as the primary or predisposing cause, (2) a sudden perturbation as a second and additionally synchronizing cause, acting as a trigger in the senescing life-stage, and (3) biotic agents as tertiary or contributing and dieback hastening causes (Mueller-Dombois and others 1983). This concept has been broadened to include diebacks and declines of other forest trees (Mueller-Dombois and others 1983) and even of certain grasses, shrubs, and vines (Mueller-Dombois 1983), in which the dieback populations themselves occur in cohort communities of low species diversity.

Decline Concept

Manion (1981) characterized declines as diseases caused by the interaction of a number of interchangeable, specifically ordered abiotic and biotic factors to produce a gradual general deterioration, often ending in the death of trees. A similar definition has been proposed by Houston (1973, 1982, 1984), who indicated that declines are diseases initiated by predisposing effects of biotic or abiotic environmental stresses which culminate in attacks, often lethal, by organisms of secondary action. In the absence of stress, the secondary organisms are unable to attack the trees successfully. Conversely, trees under stress may recover after removal of the stress factor were it not for attack by the secondary organisms. This is not to say that severe stress, if repeated or prolonged, cannot result in tree death.

Houston (1982) recognized two phases of the decline complex—dieback and decline. He conceived dieback to be a progressive development of symptoms beginning with the dying back of buds, twigs, and branches, which often results from the effect of the stress factor(s) alone. Trees often recover once the stress abates. Decline refers to the phase in which the vitality of the entire tree lessens and often culminates in death. This phase usually results from attack by secondary organisms on stress-altered trees. Recovery from this phase is less likely to occur after stress abatement. The terms dieback and decline are often used interchangeably, as in the case of the ohia problem.

Decline diseases are difficult to diagnose. The symptomatology of most decline-type diseases is remarkably similar—rootlet mortality and general crown dieback—and can be caused by a large number of biotic and abiotic factors, either alone or in combination. Such symptomatology offers little explanation of the cause of the problem, especially where several causal factors are involved and where primary injury occurs in the root system. This symptomatology makes it difficult to determine whether all declines with similar symptoms have similar etiologies.

Also, diagnosis is difficult because decline-type diseases usually involve a sequence of events triggered by some environmental, site, or biotic factor or factors resulting in stress. Often, these "triggers" or "incitants" are ephemeral, and their occurrence may not be known at the time diagnosis is attempted. The degree of decline depends on the severity and continuity of the initial stress factor; presence of disease organisms and insects capable of attacking the stressed trees; and interrelationships of these organisms, such as their sequence of occurrence and how they are influenced by environmental and site factors.

Another difficulty in diagnosing decline-type diseases is that they generally occur in mature forest ecosystems involving large trees. Conducting controlled experiments is difficult under such conditions, especially when two or more variables are studied. On the other hand, conducting experiments using seedlings in a greenhouse or environmental chambers in which one or a few factors are easily controlled, and extrapolating the results to the forest can often be misleading.

Several factors that may be involved to some degree in the ohia decline syndrome include poor soil drainage, drought, nutrition, toxic elements or compounds, intraspecific competition, senescence, insects, and pathogens. Site conditions or environmental factors may be the primary factors in the initiation of ohia decline (Papp and others 1979). This would support Houston's (1973) concept of decline diseases—tree mortality results from a sequence of events that starts with stress, which predisposes trees to attack by organisms, which eventually kill them. In the case of ohia decline, the organisms involved are primarily *P. bilineatus* and *P. cinnamomi*. A biological evaluation of ohia decline generally supports this assessment (U.S. Dep. Agric., Forest Serv. 1981).

Mueller-Dombois (1983), however, disagreed with the above conclusions as well as with the decline disease concept of Manion (1981). Instead, he proposed that senescence is the

Figure 11—Severely declining ohia forest on pahoehoe flow surrounded by healthy forest on a'a flow.

Figure 14—Soil typically associated with some Wetland Dieback stands shows a thin gley horizon at the pahoehoe lava subsurface contact and an overlying layer of rhizomes and organic muck.

Figure 12—Severely declining ohia forest showing some surviving trees at least 15 years after estimated onset of decline.

Figure 15—Ohia Displacement Dieback is characterized by an almost continuous understory of healthy treefern.

Figure 13—Healthy ohia reproduction under stand of ohia in severe decline of Wetland Dieback type.

Figure 16—Small area of dead ohia ("hotspot") in otherwise dense, healthy ohia forest. Trees died synchronously over a relatively short time and the original affected area has not expanded.

Figure 17—Profile of rugged surface jumble of a'a lava shows organic material and small basalt clinkers. Such soils are well drained and usually support healthy ohia but also are found in Pubescent Dieback.

Figure 18—Adult female two-lined ohia borer *(Plagithmysus bilineatus)* depositing eggs in bark crevices of ohia.

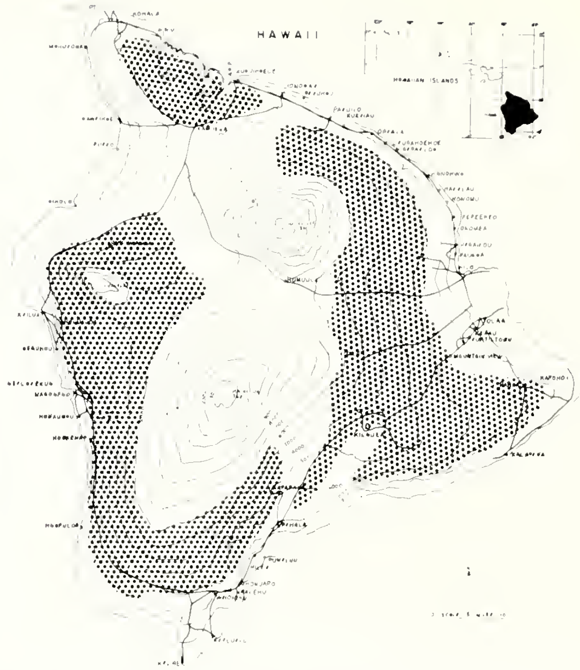

Figure 19—The two-line ohia borer *(Plagithmysus bilineatus)* is widely distributed on the island of Hawaii.

Figure 20—Spiraling larval gallery of *Plagithmysus bilineatus* on ohia effectively girdles the trunk.

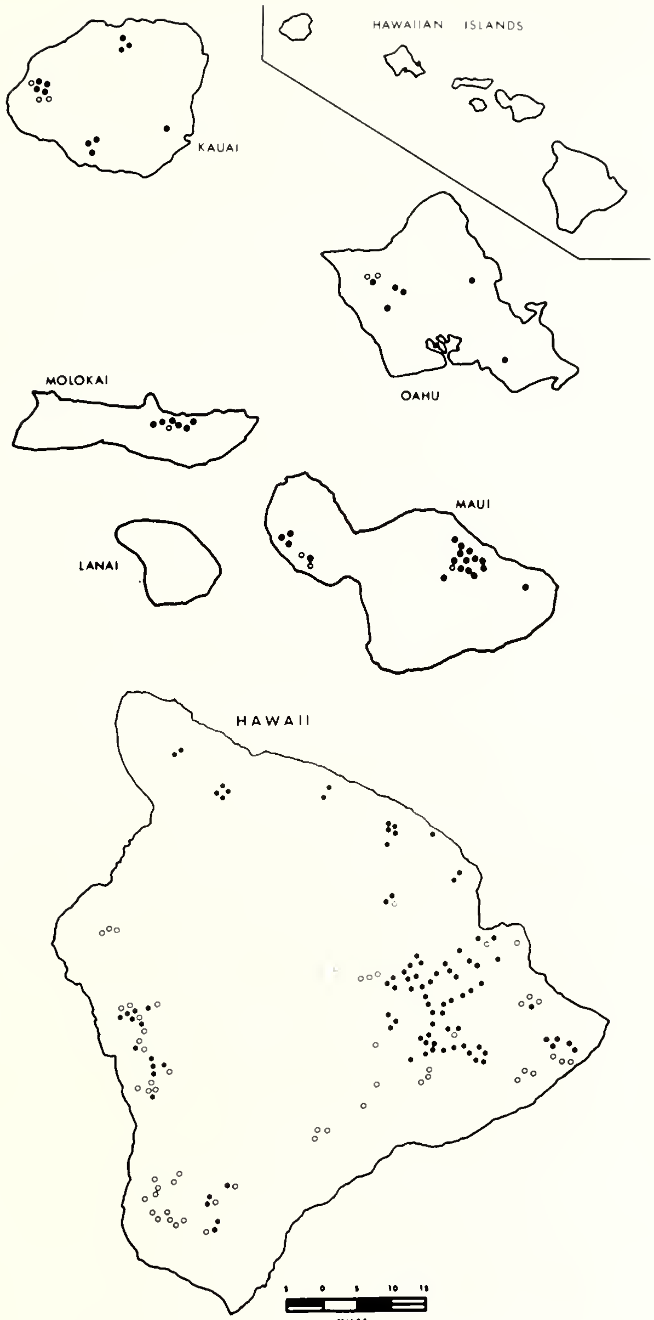

Figure 21—The fungus *Phytophthora cinnamomi* is widely distributed in the Hawaiian Islands.

predisposing factor, and—because senescence in plants is a natural phenomenon—conditions such as ohia decline can not accurately be termed diseases. Senescence is indeed a natural phenomenon, and all plants surviving to the end of their biological age potential go through such a period in their life cycle. However, even if all ohia trees suffering from decline were in senescence before the onset of decline (and there is no experimental evidence to support this), the fact that they are declining is due primarily to external stress factors and not to senescence *per se*. Innate tree vigor (= senescence?) at the time stress starts undoubtedly is an important factor in the typical decline syndrome, but the severity of the resulting decline is equally or perhaps more influenced by the strength and duration of the predisposing stress factors. The theory of Mueller-Dombois and others (1983) further deviates somewhat from the usual definition of senescence by including those cohorts whose life stage has not yet reached the normal biological age of senescence, but may have been induced into physiological senescence by environmental stresses. In our view such a broad definition of senescence in relation to declines in general helps little to identify the real precipitating causes of a particular decline problem and the term "synchronous cohort senescence" does not appear to describe in a meaningful way the etiology of ohia decline.

MANAGEMENT OF DECLINE AREAS

Because of the generally poor form of most trees and wood characteristics that make it unsuitable for most commercial purposes, ohia has little commercial value. For this reason, commercialization is not an important factor to be considered in managing areas affected by ohia decline. The ohia forests are, however, important for watershed protection, as a unique ecosystem valued for esthetic purposes, and as a habitat for large numbers of endemic flora and fauna—some of which are rare and endangered. The following comments concerning management needs are addressed to these functions.

Watershed Protection

The watershed characteristics of ohia decline areas on Mauna Loa and Mauna Kea differ significantly. On Mauna Loa, sites vary from bare recent lava flows, to shallow organic soils overlying older flows, to deep soils derived from ash. Even though most of the soils in the decline area are poorly drained, all the precipitation the area receives eventually percolates into a subsurface water system. There are no established stream channels, although during periods of high rainfall surface flow is considerable. Even though the ohia canopy has been reduced over much of the area, subcanopy cover remains high. Following decline, dense stands of young ohia have developed in many areas. Because of this dense cover and the drainage characteristics of the site, ohia decline has

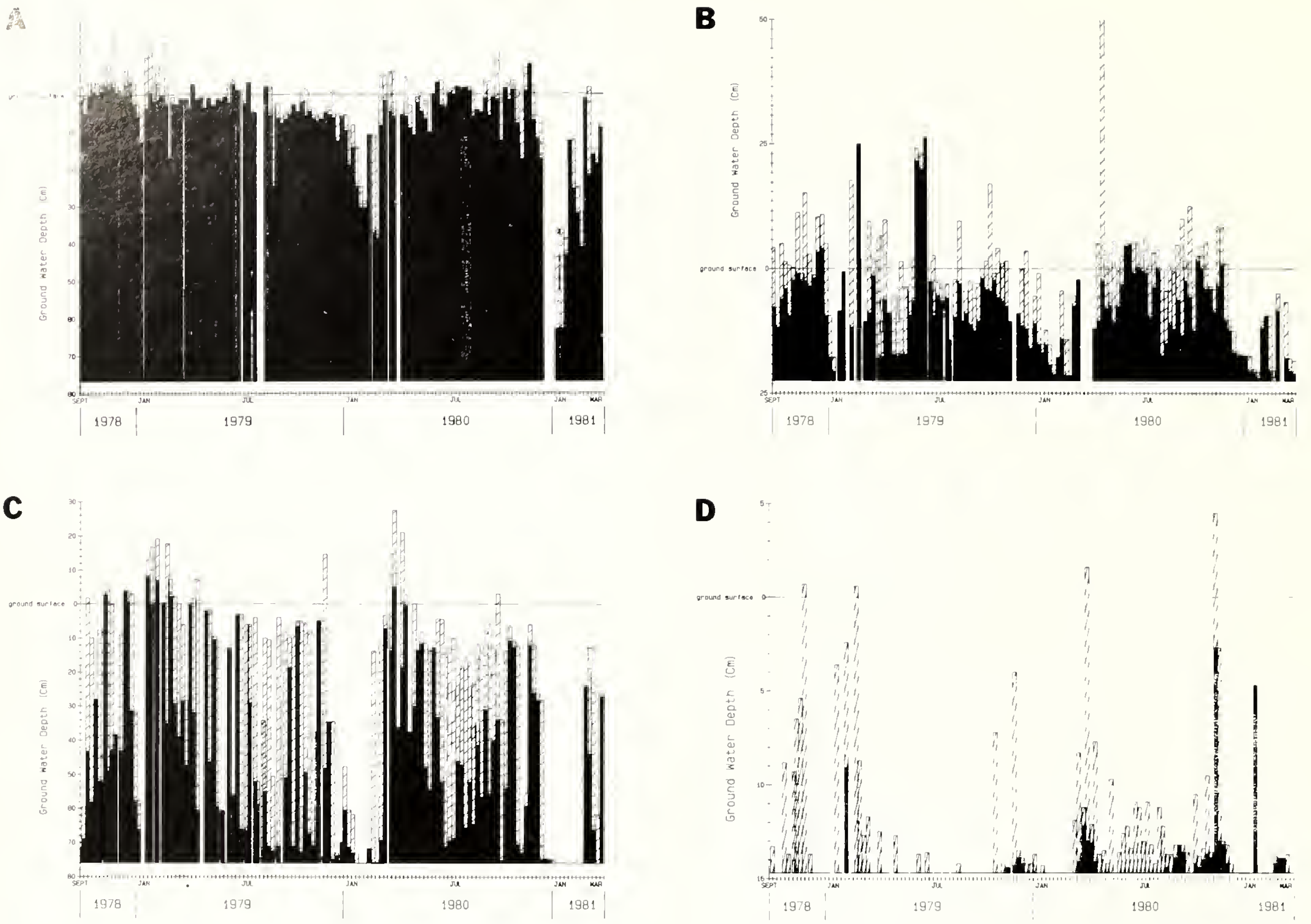

Figure 22—Ground water levels of four major drainage categories as related to ohia decline structural types: (A) very poorly drained—Bog Formation Dieback, some Wetland Dieback; (B) poorly drained—Wetland Dieback; (C) moderately well drained—Ohia-Koa Dieback, some Ohia Displacement Dieback; and (D) well drained—healthy stands, Pubescent Ohia Dieback, Dryland Dieback. Shaded bars represent weekly high water levels, and solid bars represent observed water levels.

had no adverse effect on the watershed values of this area, and none are foreseen. From this standpoint, therefore, no remedial action is necessary.

The situation on Mauna Kea is different. Soils are older and—for the most part—derived from ash deposits. Several permanent stream channels traverse the area and carry runoff water to the ocean a few miles away. The ohia canopy as well as the subcanopy species have been greatly reduced. Treeless bogs of various sizes covered with grasses, sedges, and sphagnum are being formed. Drainage has been greatly impeded in some areas through the formation of hardpans in the soil, and surface flow is considerable, which sometimes results in sheet erosion. Even with these deleterious effects of decline, there has been no major impact on amount of stream runoff or increase in stream sediment load, mostly because lost vegetation has been replaced by dense cover of *Dicranopteris* ferns and bog vegetation. Areas bare of plant cover are rare. Measurements of runoff and sediment loads from streams draining the Mauna Kea decline areas should be continued. However, unless these measurements indicate a significant deterioration of the watershed, no remedial steps need be taken.

Parts of the similar Maui decline area were replanted to *Eucalyptus robusta* and *Melaleuca quinquenervia* in the early 1900's. Current survival and growth of these two species are excellent. Soil moisture levels in the plantings are also much lower than in the adjacent nonplanted areas. Skolmen (1984) has also obtained good survival and growth of these two species, as well as of *Alnus nepalensis*, on a bog area on Hawaii after 2 years. If it is deemed necessary to replant the Mauna Kea decline area in the future because of further deteriorating conditions, these species should do well. Some concern exists, however, that *M. quinquenervia* might spread by natural seeding into other areas. This has occurred in the Maui decline area to a limited extent. Because of the value of Hawaiian forests as laboratories for the study of speciation and natural history in general, we recommend the use of introduced and potentially disruptive species only if danger of watershed deterioration is significant.

Ecosystem Values

Unquestionably, decline has severely affected the ohia canopy over a large area on the island of Hawaii. Impact on the subcanopy vegetation has been variable but generally less. Although predecline data are not available, some rare plant species may have been completely lost. Native bird populations in some areas have been significantly reduced, followed by an increase in some introduced species.

Extensive studies of post-decline succession clearly show, however, that most of the vegetation now covering decline areas is composed of native species. Although several introduced species have invaded some areas, these species apparently have had surprisingly little impact on recuperation of native species, including ohia. Many of the shade-intolerant introduced species will probably disappear with canopy closure. The major exception is the rapid invasion of some areas by *Psidium cattleianum,* especially along peripheral areas such as those adjacent to cane fields, roads, and other areas of human disturbance. No feasible means of controlling this plant is available. Further spread could be slowed, however, by reducing the population of feral pigs in the forest, which are important in seed distribution.

Feral pigs also are responsible for the establishment of other introduced plants. Grasses such as *Setaria palmifolia* colonize bare areas formed by the pigs' rooting activities, which also directly destroy seedlings of many native species. For these reasons, reduction of feral pig populations in decline areas would probably result in more rapid recuperation of native vegetation.

The future course of successional events in ohia decline areas on the island of Hawaii is uncertain. The evidence presented in this report, however, points to continuation of an ecosystem dominated by native plants, but one that will differ—radically in some places—from that present before decline. With the exception of attempts to control invasion and spread of introduced plants and to reduce the population of feral pigs, little can be done on a practical basis to speed the recovery process.

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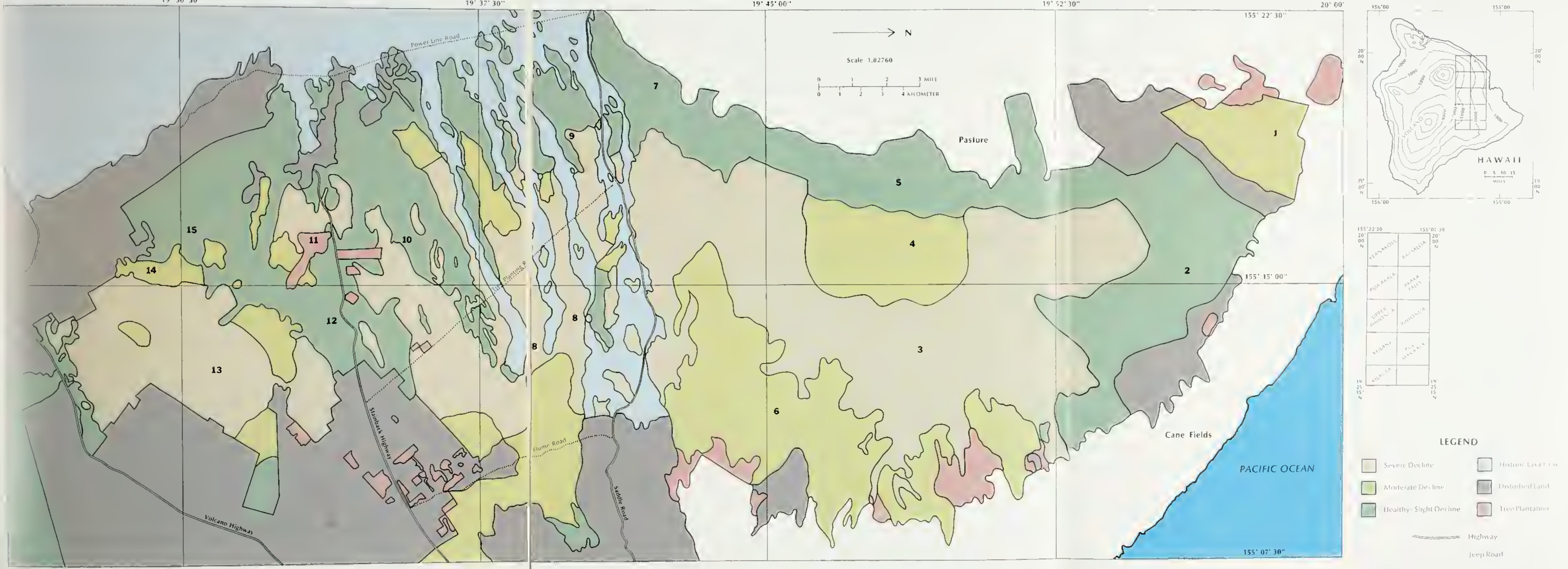
Distribution and Severity of
OHIA FOREST DECLINE
Island of Hawaii



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Portions of the ohia *(Metrosideros polymorpha)* forests on the windward slopes of Mauna Loa and Mauna Kea on the island of Hawaii began dying in 1952. Little mortality has occurred since 1972. About 50,000 ha are affected by the decline. Individual trees exhibit several symptoms, from slow progressive dieback to rapid death. Seven types of decline have been identified on the basis of differential response of the associated rainforest vegetation. Two of the types, Bog Formation Dieback and Wetland Dieback, make up more than 80 percent of the decline area. The decline has affected bird populations and plant species in some areas, but has had no major effect on runoff or water quality. Ohia decline appears to be a typical decline disease caused by a sequence of events. Poor drainage is probably the major cause of stress and is followed by attack of the ohia borer *(Plagithmysus bilineatus)* and two fungi *(Phytophthora cinnamomi* and *Armillaria mellea),* which kill the trees. Except for controlling introduced plants and feral animals that spread them, little can be done to ameliorate the effects of the decline.

Retrieval Terms: Armillaria mellea, Metrosideros polymorpha, Plagithmysus bilineatus, Phytophthora cinnamomi, decline, rainforest, Hawaii

United States
Department of
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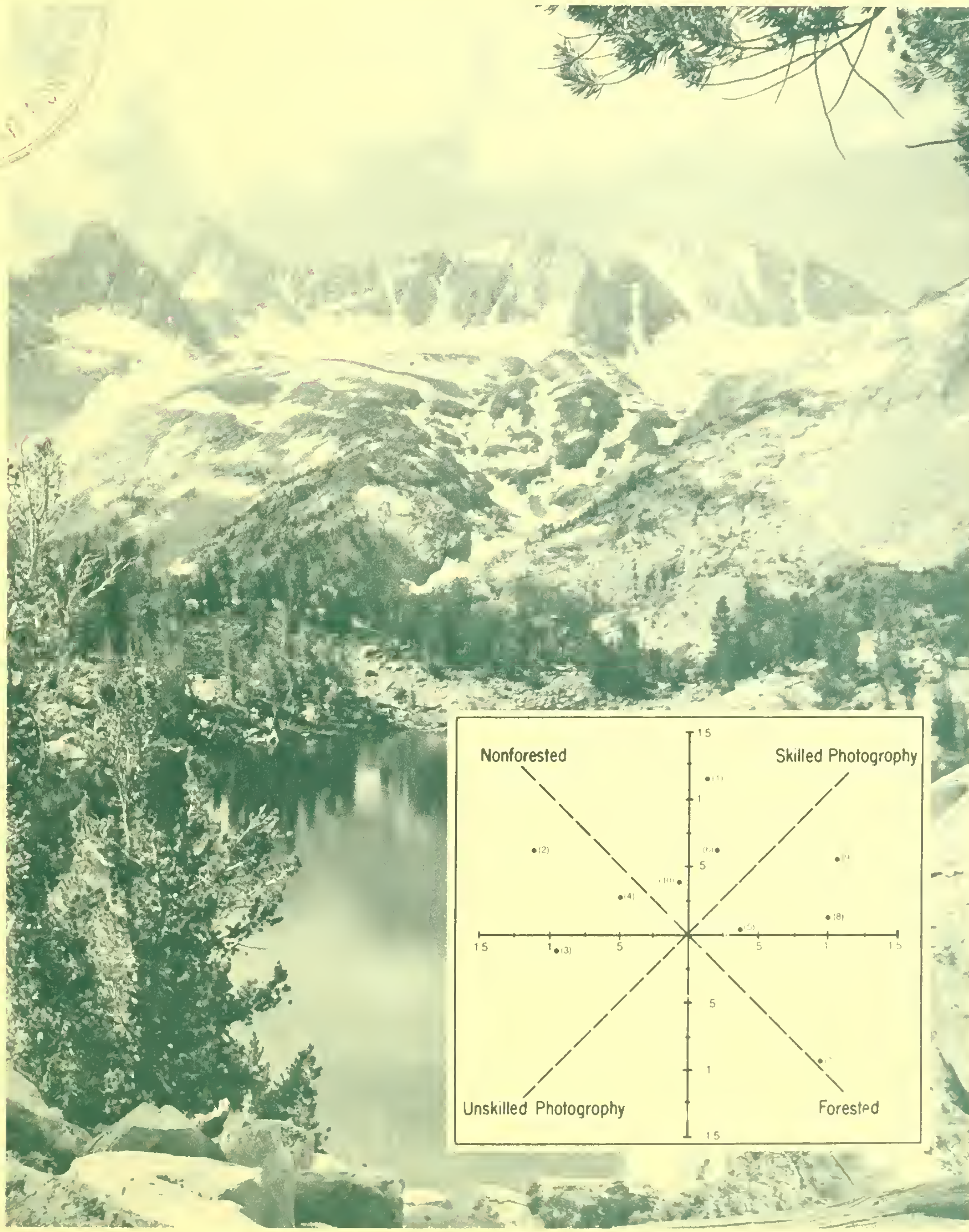
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-87

Evaluating Statistical Validity of Research Reports: a guide for managers, planners, and researchers

Amanda L. Golbeck



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Preface

Just as some managers are more professional, effective, and skilled than others, so are some researchers more competent than others in their use of statistics. And, just as managerial decisions have many dimensions, so do statistical decisions. However, decisions of a midlevel manager are subject to scrutiny of superiors only, but statistical decisions in a research report are subject to the scrutiny of all who read it. Consumers of research reports must not just accept conclusions, but must investigate the methods used to obtain them. I urge resource managers and planners to read research reports critically and to judge the choice of statistical method. In this report I explain the need to be critical and describe some ways in which conclusions can be evaluated. In addition, I recommend discussing the statistical validity of research reports with professional statisticians before applying the results to managerial or planning decisions.

Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

May 1986

Evaluating Statistical Validity of Research Reports: a guide for managers, planners, and researchers

Amanda L. Golbeck

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IN BRIEF . . .

Golbeck, Amanda L. **Evaluating statistical validity of research reports: a guide for managers, planners, and researchers.** Gen. Tech. Rep. PSW-87. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1986. 22 p.

Retrieval Terms: scaling of attitudes, statistical assumptions, ordinal data analysis, sampling biases

Inappropriate statistical methods, as well as appropriate methods inappropriately used, can lead to incorrect conclusions in any research report. Incorrect conclusions may also be due to the fact that the research problem is just hard to quantify in a satisfactory way. Publication of a research report does not guarantee that appropriate statistical methods have been used, or that appropriate methods have been used correctly. Publication also does not guarantee that actual measurements are reasonably close to the underlying concept.

You may not be able to tell if an appropriate statistical method was used correctly. You can, however, (with the help of a professional statistician) judge whether the choice of method was appropriate. You can also judge how close the actual measurements seem to the underlying concept that the researcher is studying.

Two steps are preliminary to judgments about whether the choice of method was appropriate. The first involves categorizing the study according to its primary statistical purpose in terms of examining variables. This purpose may be describing variables, testing hypotheses about variables, exploring relationships among variables, or building prediction models using variables. The statistical purpose of most studies of visual quality has been exploring relationships or building models.

The second preliminary step to evaluating the validity of a research report involves categorizing variables according to their level of measurement. This level of measurement may be (1) nominal, (2) ordinal, (3) interval, or (4) ratio. Most studies of visual quality involve several variables with different levels of measurement. One of these variables is usually an attitude or preference variable having an ordinal level of measurement.

Any given statistical technique presumes a specific level of measurement for the variable(s). Use of a statistical technique upon data that are at a lower level of measurement than what the technique presumes leads to results that are neither empirically nor semantically meaningful. Many visual quality researchers controversially have used statistical techniques that presume interval level measurements upon attitude or preference variables that have ordinal (lower) levels of measurement.

A variety of methods can be used to scale an attitude variable (such as Likert scaling and paired comparisons), all of which result in an ordinal level of measurement. Some researchers make psychological assumptions (e.g., invoke Thurstone's Law of Comparative Judgment) to claim that they have achieved an interval level of measurement for their attitude variables. If you are not willing to accept psychological theory as an ingredient to the determination of level of measurement, then you take the same position that professional statisticians do: no feasible method is available for deriving interval data from ordinal rating scale data.

When the statistical purpose of the study is that of exploring relationships among variables, several methods are available, including simple correlation, factor analysis, and multidimensional scaling. Several measures of simple correlation are available, each presupposing a certain level of measurement for the two variables to be correlated. For hypothesis testing purposes, factor analysis presupposes an interval level of measurement and some types of multidimensional scaling presuppose an ordinal level of measurement.

When the statistical purpose of the study is to build prediction models using variables, the most commonly used methods are regression methods. These typically use a straight line to approximate the relationship between one dependent variable and one or more independent variables. For hypothesis testing, linear regression requires that variables meet five assumptions: fixed, independent, normal, equal variance, and linear. Standard linear regression methods presuppose an interval level of measurement. When the variables are nominally or ordinally scaled, log-linear models should be used in the place of standard linear regression methods.

Inappropriate sampling methods can also lead to incorrect conclusions in any research report. The planned introduction of chance or probability into a sampling method can minimize or eliminate the possibility of bias. Small convenience samples—consisting only of students, for example—cannot yield valid measures of general public attitudes and preferences in the area of visual quality or in any other research area.

INTRODUCTION

The dependency of conclusions upon the choice of statistical methods can be illustrated by an example. The Ecological Society of America surveyed members, applied certain statistical methods to the data, and concluded that "applied ecologists and other ecologists were in remarkable agreement" in most of their views on publishing (Ecological Society of America 1982, p. 27). But, by applying different statistical methods to the same survey data, another researcher arrived at the opposite conclusion: applied ecologists and other ecologists differed significantly in their views (Saunders 1982, p. 336).

The fact that publication of research results does not assure their correctness can be illustrated by another example. Schor and Karten (1966) studied the statistical methods used in a large series of medical studies reported in several journals, and found that only 28 percent of them were statistically acceptable. This finding led the American Statistical Association to raise the question of whether a code of principles can be maintained to assure basic levels of statistical competence, or whether formal certification is necessary to assure credibility of an author. At this writing, the question has not been resolved, and levels of statistical training and competence vary among the users of statistics.

In fields such as visual quality where the concepts in question are sometimes harder to quantify than those in medical research, and where less professional statistical input is employed, the problem is likely to be worse than that reported by Schor and Karten (1966). The fact is, despite its numerical base, statistics is art as well as science. Often the user of statistics must choose among methods, a somewhat subjective process, and may use the method in a subjective fashion.

This subjectiveness invalidates neither statistics as a science nor statistical methods. But, nonstatisticians should be aware of this "artistic side" of the discipline. Do not unthinkingly trust figures that are published, posted on a bulletin board, or used for political purposes, the way that—for example—statistics for cost-of-living and unemployment are sometimes used. Carefully examine how figures were derived before believing them.

Sections such as the abstract and management implications in reports are convenient. But beware! Don't accept the conclusions until you have investigated the appropriateness of the analytic methods. You can examine the methods section of a research report and judge the choice of statistical method; however, often it is difficult to tell if an appropriate method was used correctly. If you don't feel competent to judge the appropriateness of the analytic methods, the best thing to do is have some doubts, and ask a professional statistician for help.

This report shows the need to judge the statistical validity of research results, especially those involving many variables or theoretical concepts. It explains—at a level of complexity compatible with the statistics involved—how methods can be evaluated. This report is a statistical guide for resource managers and planners, as well as for physical and social scientists, to use while reading research reports. It should also prove useful to researchers in planning, conducting, and reporting their studies.

1. IDENTIFY VARIABLES

The fundamental element of statistical thinking is a *variable*. Defined in simplest terms, a variable is the object of interest that is measured or counted. It can be age of mother at birth of a child, decay time of an isotope, frequency of lung cancer, angular width of a panoramic scene, density of trees in foreground, or practically any numeric quantity.

Frequently the object of interest is a theoretical concept that is not directly measurable, scenic beauty, for example. In such a case, much of the basic research is devising methods of measurement. Breakthroughs in knowledge often occur when a measurement procedure is developed or discovered that allows previously unmeasurable theoretical concepts to be quantified.

Two kinds of definitions are used in research: theoretical and operational. Researchers think with theoretical concepts. They conduct empirical research using operationalized concepts.

A *theoretical definition*, like most ordinary definitions, defines a concept in terms of other concepts which supposedly are already understood. In this type of deductive system, certain concepts are undefined or primitive, and all other concepts are defined in terms of these. An example may be

taken from Euclidean geometry, where the concepts of point and line are undefined. Other geometric concepts such as triangle and rectangle can be theoretically defined in terms of these fundamental concepts. An example relevant to visual quality is the concept of scenic beauty. A theoretical definition might be a natural scene that is pleasing.

Operational definitions of concepts include procedures for classifying and measuring them. The concept of scenic beauty has various operational definitions. A common one is "the quantification of public preferences for certain formal aspects of a landscape (e.g., color, line, form)."

This definition may be an imperfect indicator of the underlying concept. Operational definitions are, for this reason, often considered as indices. Three assumptions are implicit in the typical operational definition of scenic beauty: (1) the esthetic quality of a landscape is meaningfully correlated with certain preferences for that landscape, (2) those preferences are those of the general public, and (3) the esthetic quality of a landscape can be described in terms of formal aspects only (e.g., forms, lines, textures, colors). These assumptions clarify that the operationalized concept in this research is public preferences for certain formal aspects of a landscape (or, more directly, of a photograph), which may not be equivalent to the theoretical concept of scenic beauty (Carlson 1977).

Conclusions arising from quantitative research apply strictly to concepts as operationally defined. Propositions involving theoretically defined concepts cannot be empirically tested. The question arises whether, in practice, a particular operational definition is reasonable. Generally, is there any logical way to determine if an operational definition adequately measures the theoretically defined concept? Most researchers do not believe so. Instead, they rely on simple convention or general agreement that a given operational definition should be used as a measure of a certain concept. Such convention or agreement is based on the argument that the operations "seem reasonable" on the basis of the theoretical definition. That is, the operational definition seems reasonably close to the underlying concept.

The problem can and does arise of then having several different operational definitions or indices associated with each theoretical concept, each of which may produce significantly different results. For example, if there are two distinct operational definitions of scenic beauty or landscape preferences, two distinct hypotheses are being tested. Researchers may have to revise or clarify the theoretical definition when several scientifically acceptable operational procedures carried out under similar circumstances yield different results.

Recognizing the difference between a theoretical concept and its operational definition is largely a matter of common sense. When you read reports of studies on a theoretical concept like scenic beauty, ask these questions:

- What was studied?
- Can the concept(s) be measured directly? If not, what was actually measured?
- What assumptions had to be made to get back to the underlying concept?
- Are the assumptions acceptable?

2. DETERMINE STATISTICAL PURPOSE

Studies in any field have both theory and methods components. Both can be manipulated to support a desired conclusion. In terms of theory, a researcher may tend to be selective and may report only references supporting a favored position. Some researchers may also manipulate the conclusion by publishing only the supporting results. In terms of methods, the match between the research problem and the statistical method used may not be good, or appropriate statistical methods may have been used incorrectly.

Many statistical methods are available for describing and analyzing variables. Depending on the type and number of variables and the problem at hand, some statistical methods are much more satisfactory than others in producing a reliable conclusion. Before examining the conclusions of a study and the theory they support, try to categorize the statistical procedures, to get a sense of the plausibility of conclusions. The primary purpose of the statistical methods can be one of these:

- Describing variables
- Testing hypotheses about variables
- Exploring relationships among variables
- Building prediction models using variables.

The majority of studies of visual quality have done more than merely describe variables. In particular, a few studies have tested hypotheses about variables, and even more studies have either explored relationships among variables or built prediction models using variables.

2.1 Describing Variables

Descriptive statistics can be used to organize and summarize data. Such techniques help both researchers and readers of research reports to understand more readily the importance of the data.

The researcher begins with *raw data*. These are the values that are collected for each variable, unaltered by statistical or other manipulation. They are obtained by counting or measuring with a scale. For example, a sample of 100 people is taken to yield 100 values on the variable, scenic quality. These 100 observations or measurements are raw data.

The purpose of descriptive statistics is to examine the distribution of values for single variables in order to gain understanding of the research problem. The researcher attempts to condense the values (data) for a variable to a few representative numbers. For example, it is difficult for the researcher to comprehend the relation of 100 individual values for scenic beauty to the research problem. Therefore, some descriptive statistics can be computed to reduce the 100 values to one or two convenient summary measures.

Descriptive statistics can answer four crucial questions about a data set:

1. Where do the bulk of values fall? For example, how do most of a sample of observers rate the scenic beauty of a particular landscape?

2. What proportion of the values fall in a range of particular interest? For example, what proportion of the observers gave the landscape a positive scenic beauty evaluation?

3. What are the upper and lower extreme values? For example, what is the highest and lowest scenic beauty rating given to a particular landscape within a sample of observers?

4. What is the relationship of a particular value to the group of values? For example, do the sample of observers have widely differing evaluations of the beauty of a landscape?

One common type of representative number is a measure of location or central tendency. This number indexes the center of a distribution of a set of observed values for a variable. The most common measures of location are the *mean* (arithmetic average), the *median* (the value that divides the ordered observed values into two groups of equal size) and the *mode* (the value that occurs most often).

Measures of dispersion are also common. These measure the variability among values for a variable. Three common ways of measuring dispersion are the *range* (largest value in a set of observations minus the smallest value), the *sample variance* (sum of squared deviations of each observation from the mean, divided by the number of observations minus 1), and the *standard deviation* (square root of the sample variance). The latter two measures show how tightly packed the observations are about the mean.

Beware of the seductive ease of summary statistics. Many individuals, especially social scientists, are lured to "invent" new types of summary statistics for their research problems. Summary statistics can mask important differences within and between groups of subjects. For example, these two sets of measurements have the same mean (=20)

21, 22, 19, 18

1, 2, 3, 74.

These two sets of measurements are very different, but the reported summary statistic does not reflect this.

If a study was mainly descriptive, one or more of these measures will have been the statistical focus: mean, median, mode, range, sample variance, and standard deviation.

2.2 Testing Hypotheses

Whenever numerical results are subject to chance, the researcher can go beyond descriptive statistics in analyzing data. *Statistical inference* uses statistical methods designed specifically to assess the likelihood that research results are explainable by chance.

One type of statistical inference involves testing a *statistical hypothesis*. A statistical hypothesis is a statement concerning the distribution of probabilities for different values of a random variable. A *random variable* is a variable that has probabilities attached to specific numeric values. That is, there is a certain probability that a specific value will occur if only one observation is taken. For example, suppose the variables is preference for a particular type of landscape, and it can take on the value 1 (indicating low preference), 2, 3, 4, or 5 (indicating high preference). Preference would be a random variable because probabilities are associated with each possible value. Thus, if only one observation is obtained for the variable, the probability that the value equals 1 (is low) exists.

The researcher may hypothesize that the preference variable has a uniform distribution. That is, the probability of taking on a specific value is the same for all five values of the variable, and the probability is equal to 0.2. The latter statement is a statistical hypothesis. The researcher then asks each person in a sample to rate their preference for the landscape. These (raw data) are then summarized into a relative frequency distribution, which indicates the proportion of people in the sample that gave a preference rating of 1, the proportion that gave a rating of 2, etc.

Testing compares the observed relative frequency distribution with the hypothesized probability distribution and answers the question: Do the relative frequencies differ significantly from 0.2? Two hypotheses are tested at a time: the *null hypothesis* and the *alternative hypothesis*. In the example, a null hypothesis would be that preferences are uniformly distributed, and the alternative hypothesis would be that preferences are *not* uniformly distributed. A *hypothesis test* is used either to reject or not reject the null hypothesis while knowing the probability that the decision is wrong (probability of error).

As can be seen from the example, the null hypothesis is an educated guess that the distribution of the observed values shows no basic difference from the assumed probability distribution. Not rejecting the null hypothesis then means that the data show no systematic difference from the assumed distribution, that is, no difference beyond that attributable to random variation. Rejecting the null hypothesis lends support to the alternative hypothesis. In such a case, indications are that a systematic difference exists between the observation and a theoretically derived standard. That is, a difference exists beyond that attributable to random variation.

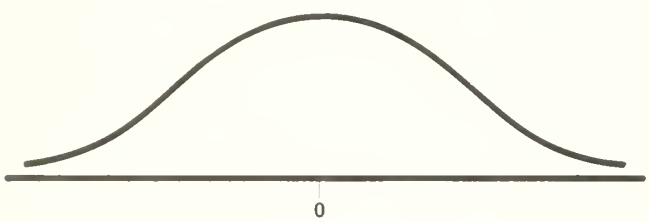
Statistical tests have four possible outcomes. Two are correct actions and two are possible errors. Consider an example concerned with landscape management, where the null hypothesis is that a road is visually subordinate to the characteristic landscape. The four outcomes are illustrated below:

Action \ Reality	Null hypothesis is true (i.e., alternative hypothesis is false): The road is in fact visually subordinate to the characteristic landscape.	Null hypothesis is false (i.e., alternative hypothesis is true): The road does in fact visually dominate the characteristic landscape.
Reject the null hypothesis (i.e., accept the alternative hypothesis): the sample indicates that the road visually dominates the characteristic landscape.	Error	Correct action
Accept the null hypothesis (i.e., reject the alternative hypothesis): the sample indicates that the road is visually subordinate to the characteristic landscape.	Correct action	Error

Certain assumptions must be true for statistical tests to be appropriate. The nature of these assumptions depends on the particular test. If appropriate assumptions are not met, the results of statistical tests are invalid.

Statistical hypotheses can be tested in either a *univariate* or *multivariate* situation. Univariate situations involve only one variable; multivariate situations involve a number of variables operating simultaneously. Visual quality research usually involves multivariate situations. The assumptions necessary for multivariate tests are usually more difficult to satisfy than are those for univariate tests.

The F-test is an example of one of the statistical tests that has appeared in the literature on visual quality. To use the F-test, three assumptions must be met. When you see an F-test reported for research on visual quality, look for some evidence within the article that the assumptions described below are true of the data used (often you will be given no evidence—in that case, you are simply unable to judge whether the assumptions are met). (1) Each variable used in the test has its own underlying normal (i.e., bell-shaped) probability distribution.



The mathematical form for the *normal probability distribution* is given in the *glossary*. (2) The population variance of all variables used in the test is the same. (3) The values for a random variable are *statistically independent*. In a loose sense, the last assumption means that the value obtained for one observation does not affect the values that are likely for other observations. For example, if a random sample of people were asked to rate the scenic beauty of a landscape, the ratings between people are likely to be independent. But if a random sample of people were asked to give such a rating before and after a landscape intervention, then for each person the two ratings are not independent. Analyses using statistical inference are occasionally reported for visual quality research. The difficulties of conducting visual quality research, together with its multivariate character, seldom permit valid use of inferential statistics.

Knowing the names of three common tests—t-test, Z-test, F-test—will help you recognize when the statistical purpose of a study is testing statistical hypotheses about variables.

When reading reports of studies that statistically tested hypotheses, ask these key questions with the help of a professional statistician:

- What statistical assumptions are necessary for using that test statistic?
- Are these assumptions either true or closely approximated by the data?
- At worst, do the assumptions seem reasonable for the given set of data and for the study approach to this particular problem?

2.3 Exploring Relationships

Most data sets involve observations associated with more than one aspect of a particular background, environment, or experiment. Because of this, data are usually multivariate, as in visual quality research. The basic question in the multivariate situation is the following: If a large number of variables are characterized by complex relationships, what will make the problem easier to understand?

Several statistical methods are available to simplify a multivariate situation. One of the simplest methods begins with the concept of *association*. Two variables are highly associated if the value of one variable can be used to reliably predict the value of the other. For example, the researcher might find that distance to the back ridge was highly associated with scenic beauty rating. This could mean that either the closer an observer is to the back ridge, the higher the scenic beauty rating (positive association); or the closer an observer is to the back ridge, the lower the scenic beauty rating (negative association).

The different statistical measures of association—such as Pearson's product-moment correlation, Spearman's correlation coefficient, joint biserial correlation—are all summary measures and must be interpreted cautiously like any single number that summarizes an entire set of observed values. Also, certain assumptions must be met for a particular measure of association to be appropriate.

In the multivariate research problem, the researcher usually has a large number of variables. Describing the interrelations among them could go beyond the concept of simple association between two variables. In particular, the researcher could (1) study the associations of a large number of variables by clustering them into groups within which variables are highly associated; (2) interpret each group by studying the variables in it; and (3) summarize many variables by a few *post hoc* variables constructed to represent each group.

Several statistical techniques can be used to accomplish the above three goals. In general, these techniques reduce a complex data set into something that is easy to understand, visualize, and interpret, while retaining sufficient detail for adequate representation. They focus attentions on meaningful relationships between variables and uncover the hidden structure of a data base. Two techniques for exploring relationships among variables are factor analysis and multidimensional scaling. For more details, see *chapter 5*.

2.4 Building Prediction Models

No empirical problem is directly concerned with mathematical (in this case, formal probabilistic) concepts. The researcher needs to translate an empirical problem into formal probabilistic terms before using probability theory to analyze the problem. This translation amounts to building a *probability model* of the problem.

There are many different ways to build a probability model. Choosing the model that best fits the data is straightforward when the adequacy of a particular model can be tested empirically. But just as often it cannot. When a model cannot be tested empirically the researcher is forced to rely on intuitive judgment about the adequacy of how probability model components correspond to the phenomena being studied. Once again, the situation is one of "art plus science." An analysis of a problem based on a given model applies to the model, not necessarily to the phenomena. More precisely, the amount of correspondence between a mathematical description and the phenomena depends on the adequacy of the model. The model may be accurate but be impossible to use because of the difficulty of mathematical analysis. Alternatively, the model may work but be too simple to adequately represent the problem.

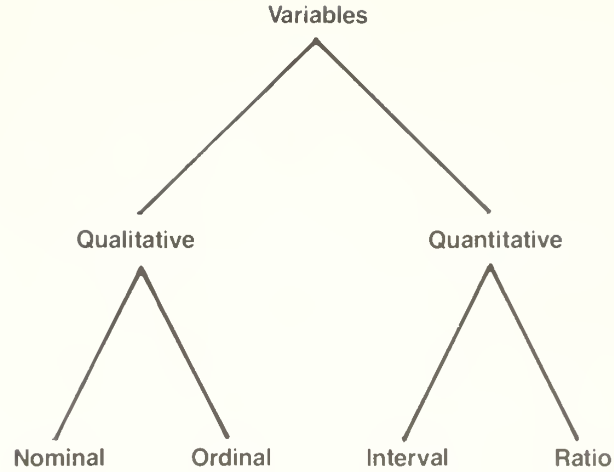
The researcher often can use regression techniques to build a linear mathematical model. Sometimes the model will be unrealistic, but acceptably so. In other words, it will give predictions that are not entirely accurate, but yet accurate enough to be useful. Such models aid in choosing the most salient group of variables and in understanding the interactive effects among them. Regression techniques for prediction of scenic beauty are discussed in detail in *chapter 6.*

3. COMPARE LEVELS OF MEASUREMENT AND ANALYSIS

To critically evaluate a research document in the field of visual quality, you need to appreciate the concept of level of measurement and the associated notion of types of variables. This is necessary because any given statistical technique presumes a specific level of measurement.

3.1 Precision of Measurements

In general, variables are either *qualitative* or *quantitative.* In turn, qualitative variables are either *nominal* or *ordinal,* and quantitative variables are either *interval* or *ratio.*



Each type of variable is a distinct level of measurement with statistical procedures that are also distinctly appropriate.

A quantitative variable measures things that are expressed as real numbers and have real-world (physical) counterparts. Examples would be temperature measured in degrees Celsius, the number of trees in a particular section of forest land, or material wealth in dollars. Qualitative variables are extensions of the concept of measurement that include certain categorization procedures ordinarily used in the social sciences. Examples would be species of trees, types of roads, or amount of vegetation cover measured in the three simple categories of low, medium, or high.

Nominal variables result from sorting things into homogeneous categories. An example would be 20 photographs of 20 individual trees sorted by species. The result might be one category of oaks and one category of pines, which could be labeled by an arbitrary number instead of by name (e.g., 1 = pines, 2 = oaks). This is the simplest level of measurement. No assumptions are made about relationships between categories. As long as the categories are exhaustive (include all the photographs) and do not overlap (no photographs in more than one category), the minimal conditions are met for the application of certain statistical procedures.

Ordinal variables also result from sorting things into homogeneous categories, but the categories are also ordered or ranked with respect to the degree, intensity, or amount of something. An example is 20 photographs of 20 individual trees categorized by estimated age of trees. With five age categories, some of the photographs would fall in category 1 (youngest), some in 2, some in 3, some in 4, and some in 5 (oldest). One point needs to be understood about the ordinal level of measurement: it supplies no information about the magnitude of the differences between the categories. Ordinal measurements do not tell if the trees in category 5 were five times older than those in category 1, or two times older, or any other information about how many years of age were represented by each category of trees. The implication of this point will be discussed in the next section.

Interval and ratio scales differ from the other levels of measurement in that they both rank observations and indicate the exact distance between them. This is the true interval

level of measurement, which requires establishing some physical unit of measurement as a common standard. Examples are length measured in meters, time measured in seconds, and age of trees measured by cutting them down and counting the annual rings of each.

Ratio scales or levels of measurement differ from interval levels only in that ratio scales allow the location of an absolute or nonarbitrary zero point on the scale. Interval scales allow the arithmetic operations of addition and subtraction. Ratio scales go further to allow comparison of values by taking their ratios. The distinction between interval and ratio levels of measurement, however, is largely academic. Most real-world examples of interval scales are also ratio scales.

3.2 Sophistication of Analysis

Research methods and operational definitions determine a level of measurement. Then, statistical procedures are applied to what is measured. Any given statistical technique presumes a specific level of measurement. The more advanced the level of measurement, the more sophisticated the statistical techniques available.

Level of Measurement

Increasing precision of analysis ↑

- **Ratio (most advanced)**
- **Interval**
- **Ordinal**
- **Nominal (simplest)**

It is always legitimate to use analysis techniques that presume levels of measurement that are one or more levels below the data. For example, interval level data may easily be collapsed into ordinal categories or ranks, and an ordinal level statistical procedure may be applied. The reverse, i.e., using an analysis technique higher on the scale of measurement than the data, is statistically invalid. For example, after collection, ordinal data can in no way be upgraded to interval measurements.

Consequently, the effect of applying an interval level statistical procedure to upgrade ordinal data is unknown. This unknown effect is, in fact, a major controversy concerning quantification in the social sciences. The following excerpt regarding a survey illustrates this controversy (Saunders 1982, p. 336):

> . . . some of the statistical analyses are questionable and provide poor examples of data analysis . . .
>
> Data are usually placed in four types based on the criteria of identity, order, and additivity (Drew 1980). These four data types in order of increasing criterion properties are nominal, ordinal, interval, and ratio data. The Likert or 1 to 5 scale (1=strongly disagree, 3=neutral, 5=strongly agree, or 1=least desirable, 3=mixed or neutral, 5=most desirable) is an example of ordinal data. Inherent to this scale is the recognition that the differences between any two responses (e.g., 2 and 3, or 1 and 2) do not represent equal intervals. While ordinal data have the properties of identity and order, they lack the property of additivity, since the interval is unknown. Sokal and Rohlf (1969) refer to ordinal data as ranked variables.
>
> Because ordinal variables lack the property of additivity, the use of such central tendency measures as the mean and standard deviation is not possible. Some authors (Labovitz 1967, Nunnally 1967, and Borgatta 1968) argue that because emotions cannot be limited to five points on a scale, but rather are on a continuum, the Likert scale may be treated as an example of interval data. Interval data have the property of additivity. However, such logic does not answer the question of interval size and equality of intervals. Nunnally (1967) is a strong advocate of using the Likert scale as interval data, arguing that almost any parametric test can be applied to these data.
>
> Examination of the social and biological literature reveals acceptance of the Nunnally positions in certain journals, its partial acceptance in other journals, and its rejection in still other journals. Generally, the calculation of means for ordinal data are accepted, or at least done, because it may show data trends. However, since most response means are in the range of 2.75 to 3.75, a neutral rating says very little about the data. The same is true for the use of standard deviations, variances, and the t test. The use of frequency categories, an appropriate statistic, are more telling about the same data set. Chi-square analysis would be an appropriate inferential statistic for such ordinal data.

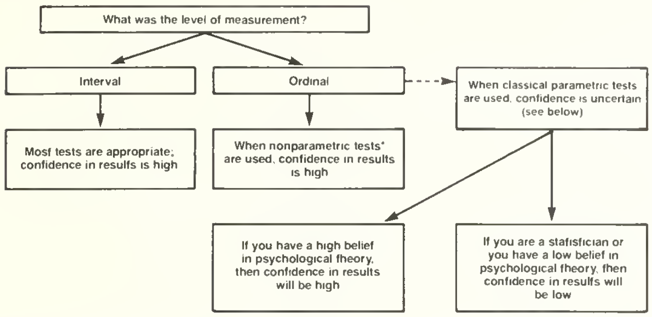
The statistical position is that using nonparametric rank-based statistical procedures (as opposed to classical *parametric statistical* procedures such as Pearson's correlation and factor analysis) for inference on ordinal variables is correct. Nonparametric procedures make no assumption of a probability model with finite numerical parameters. Classical parametric inferences should be restricted to interval level variables from the viewpoint of statistical theory, and without going into details, this position is hard to argue. Classical parametric procedures, however, are the common tools of psychological statistics. Like many areas of investigation, psychology has its own statistical peculiarities with nonstandard usage that is adapted to the prevailing practical situation. This situation in psychology is largely historical. Parametric tests were the first to be developed and still are the standard fare of introductory statistics courses. Several decades ago nonparametric tests (those appropriate for ordinal level variables) were relatively unknown to the average researcher.

Many psychologists and researchers with psychological training in environmental and other fields continue the tradition of using classical parametric statistical procedures with ordinal data. Statistical theory on the other hand dictates that using statistical procedures with an inappropriate level of measurement leads to conclusions that are neither empirically nor semantically meaningful. Unfortunately, the prevailing psychological orientation has been characteristic of the study of landscape quality and preferences.

The difference between parametric and nonparametric tests is in many cases only slight as to statistical power and statistical significance. In recent years the catalog of versatile and appropriate statistical procedures for ordinal level variables has been greatly extended to include the following (and others): median, Spearman's rank correlation, Kendall's tau, gamma correlation, Wilcoxon rank sum test, Kruskal-Wallis

test, multidimensional scaling, log-linear models, scoring based on preference pairs.

The following flow diagram may help you to judge the statistical support for or validity of a finding.



*Examples of nonparametric procedures include median, Spearman's rank correlation, Kendall's tau, gamma correlation, Wilcoxon rank sum test, Kruskal-Wallis test, multidimensional scaling, log-linear models, scoring based on preference pairs

4. QUESTION ASSESSMENTS OF ATTITUDES AND PREFERENCES

The most common characteristic of techniques used to measure (landscape) preferences and perceptions is the lack of a clearly established metric base, i.e., lack of an unambiguous interval scale. There is no general agreement on an objective, physical instrument for measuring attitudes. Without such a measuring instrument, psychological assumptions must be made in order to presume an interval level of measurement. As was discussed in chapter 3 the validity of all such assumptions is questionable.

A variety of scaling techniques have been used for the measurement of preferences and perceptions in the scenic beauty evaluation literature. Several categories of techniques include ratings and transformed ratings, rank ordering (including paired comparison), Q-sort, and the semantic differential.

4.1 Likert Rating Scales

The most common type of rating scale in the literature on scenic beauty is the *Likert Scale*. The classic Likert Scale is formed as follows: subjects are presented with a list of statements (stimuli) on a single topic. Each item on this list is intended to measure the same attitude. Subjects are instructed to respond to each statement in terms of their degree of agreement, or disagreement, usually on a scale of 1 to 5. Responses for each subject are then summed over the questions to produce a single measure of attitudes on the corresponding topic.

An example is a Likert scale consisting of 10 statements on the subject of the respondent's self-esteem (Rosenberg 1965). Three of Rosenberg's statements illustrate the idea of Likert scaling:

1. I feel that I have a number of good qualities.
2. I wish I could have more respect for myself.
3. I feel I'm a person of worth, at least on an equal plane with others.

Each of the 10 statements used by Rosenberg had the following possible responses:

(5) Almost always true
(4) Often true
(3) Sometimes true
(2) Seldom true
(1) Never true

The sum of the response scores over all 10 questions produced a single measure called the Rosenberg Self-Esteem Scale.

The Likert method scales subjects, not stimuli. Consequently, all systematic variation in responses to stimuli is attributed to differences between the subjects. The major problem with Likert Scaling is its insensitivity to the location of individual items on an underlying attitude continuum. Therefore, an absolute interpretation of a person's score in terms of that continuum is not derivable.

In Likert Scaling, the recommended set of scale scores for each favorable statement is the set of successive positive integers, e.g., 0 = strongly disagree; 1 = disagree; 2 = undecided; 3 = agree; 4 = strongly agree. For unfavorable statements, this weighting scheme is reversed, e.g., 4 = strongly disagree. If each item is scored identically in this manner and responses for all items are summed, then the possibility that each item contributes equally to the total score is maximized. The Likert Scale was developed as an improvement over rank-ordered scales by introducing intensity-scaled responses for each item.

4.2 Rank-Ordered Scales

Rank-ordered scales operate on stimulus comparison data under the assumption that the subject can rank each item in a set. The method of *paired comparisons* is one type of rank-ordered scale. Stimuli are presented in combinations of pairs, and the subject is asked to judge each pair. For example, if the goal is to determine landscape preferences by viewing 10 photographs, subjects will be shown every possible pair of photographs and for each pair will be asked which of the photographs he or she prefers. Paired comparison is based on the law of comparative judgment (Thurstone 1927): for each stimulus there exists a most frequently occurring response. That is, a subject can discriminate the relative degree of an attribute, such as scenic beauty. Further, the degree to which any two stimuli can be discriminated is a direct function of their difference in regard to the attribute in question.

4.3 Q-Sort

Q-sort is essentially a sophisticated method of rating and rank-ordering stimuli. Each subject is given a set of cards (stimuli) and asked to what extent each card characterizes the concept being evaluated. The subject is then instructed to sort the cards into a fixed number of piles in terms of the degree to which each stimulus represents the concept. These piles are taken to represent points along a continuum of representativeness of the stimuli to the topic. Determining the number of piles and the imposed frequency distribution of cards in piles is left to the investigator. The larger the number of piles the greater the potential for finer discrimination among items. An example of an imposed structure found by psychologists to be useful is the following:

Pile number:	1	2	3	4	5	6	7
Number of cards:	3	7	11	14	11	7	3

In this example the subject is instructed to place exactly 3 cards in pile 1, 7 cards in pile 2, 11 in pile 3, etc. Most statisticians recommend not forcing a distribution on the data, i.e., not predetermining how many cards should go into each pile.

4.4 Semantic Differential

The semantic differential uses sets of bipolar adjective pairs (e.g., warm and cold or hard and soft) to judge a concept. Subjects are asked to decide to what degree a concept is associated with each bipolar adjective pair. The scale for each bipolar adjective looks like a standard rating scale. Scale values are then factor-analyzed (see *chapter 5*) to answer questions regarding the number of dimensions underlying the concept. In comparison, the scaling models already discussed have all operated on the assumption of a unidimensional concept.

4.5 Two Examples of Visual Quality

A psychological orientation has been characteristic of the study of landscape quality and preferences. Two examples, one public evaluation approach and one professional evaluation approach, are given below.

4.5.1 Public Evaluation Approach

The public evaluation approach first purported to produce a separate quantitative measure of scenic beauty for a given landscape for each observer (Daniel and Boster 1976). It then produced a quantitative measure of scenic beauty for a given landscape across a sample of observers.

Data used to produce a quantitative measure for a given observer consisted of a set of preference judgments for that landscape. Specifically, the observer looked at a sample of slides taken at randomly selected locations and directions within the same area and rated each scene on a scale of, for example, 1 to 10. Furthermore, the observer was told explicitly to use the full range of the scale, if possible. The *empirical distribution function* (cumulative relative frequency) of the observer's responses was computed at each category on the rating scale. These values were taken to be estimates of the probability distribution function, i.e., the cumulative probability that the landscape will be assigned a given rating by that observer. Standard scores associated with each cumulative probability were abstracted from a statistical table of normal theory *Z-scores*, and then an average Z-score was computed for this sample distribution.

The above procedure was repeated for the same observer for at least one other landscape. Let N denote the total number of landscapes; therefore, N was greater than or equal to two. Thus, a set of N average Z-scores was obtained for the observer, one for each landscape. One of these mean Z-scores was randomly selected to represent a referent landscape. Thus, if Zbar(1) denotes the mean Z-score computed from the referent landscape, then a scenic beauty estimate for the *i*th landscape was computed by

$$\mathrm{SBE}(i) = [\mathrm{Zbar}(i) - \mathrm{Zbar}(1)] \times 100, \quad i = 2,3, \ldots, N.$$

Repeating the entire above procedure for a sample of observers allowed SBE's for a given landscape to be averaged across observers to obtain a mean SBE for that landscape. This method of estimating scenic beauty can be applied to one observer and at least two different landscapes, or to multiple observers and at least two different landscapes.

The basic scaling technique was a Likert rating scale modified for use with photographic stimuli from the same landscape (for discussion of Likert Scale, see *section 4.1*). The public evaluation approach described here attempted to improve on earlier approaches using Likert scaling by adjusting for each observer's idiosyncratic use of a rating scale, e.g., each observer's use of a different "underlying" scale.

From a statistical standpoint, this approach is nonstandard and knotty. For one thing, rating scale responses yield ordinal level measurements, which were treated as interval level measurements. The treatment is valid if each observer uses a scale of the same magnitude, that magnitude spans the entire range of response to the stimulus, and the categories are equally spaced. Because these specifications cannot be tested, they amount to psychological assumptions; the validity of results depends on the validity of these untestable assumptions.

Another problem with this public evaluation approach is that it did not follow statistical theory. By definition, the 50th *percentile* associated with a probability distribution is the specific value of a variable that corresponds to a cumulative probability of .5; the 35th percentile is the value that corresponds to .35, etc. Percentiles can be computed for any specified value of cumulative probability. The public evaluation

approach did the following for each landscape for each observer: it computed percentiles associated with a standard normal distribution for all values at which the empirical distribution function had been evaluated. But then it computed means and standard deviations of this set of standard normal percentiles. It is difficult to say exactly (or even approximately) what the resulting estimates of scenic beauty mean.

4.5.2 Professional Evaluation Approach

The professional evaluation approach used a descriptive inventory approach that relied on judgments of professionals rather than on public preferences (Sheppard and Newman 1979). The method purported to produce for a given landscape a quantitative estimate of the visual contrast created by a proposed intervention.

A detailed description of the landscape was prepared before the proposed modification. This was accomplished by breaking down the landscape into components (land and water, vegetation, and structure). Each component was described in terms of six visual elements: scale, color, line, form, texture, and space. A simulation of the landscape after the proposed intervention was then prepared, and as before, the simulation was described in terms of visual elements of landscape components. Contrast ratings were produced for landscape components to estimate the change in visual elements created by the proposed intervention. These were weighted to reflect their "relative importance," e.g., color has a weight of 3 and texture has a weight of 1. An intricate scheme was devised to produce a visual contrast rating for the landscape over all visual elements. Finally, the magnitude of the overall rating was used to determine whether the landscape intervention was approved.

Again, it is difficult to say exactly what the results mean. For one thing, I am not aware that color has been proven to be three times more important to perceptual discrimination than is texture. Where color is constant or color differences subtle, texture may prove extremely important. Also a line could be the most important visual element if it coincides with an edge (e.g., a skyline).

4.6 Magnitude Scales

4.6.1 Theory

One body of literature attempts to measure interval preferences for landscapes by using an approach called *magnitude scaling* (e.g., Daniel and Boster 1976, Buhyoff and Wellman 1980). In general, this type of approach was originally developed in signal detection theory for the ratio scaling of sensations that are physically measurable, such as heaviness of lifted weight, loudness of sound, brightness of light, and other perceptions of the five senses. Many social scientists claim that this psychophysical scaling technique solves the problem of ordinal-interval levels of measurement.

The paradigm for sensory psychophysical scaling follows. Subjects are presented with a series of sensory stimuli across a wide range (e.g., varying light intensities), one at a time in random order. Subjects are instructed to give numbers to the perceived brightness of each stimulus relative to the first light intensity, which is called the reference. So if a given light seemed 10 times brighter than the reference, the subject would give a number 10 times larger.

The results of hundreds of such numeric estimation experiments proved that humans are capable of using numbers to make proportional judgments of physical stimulation levels for virtually all of the five senses. These numeric estimates of the perceived strength of sensory stimuli were found to have a simple and regular mathematical relationship to the objectively measured stimulus values. The principle behind this mathematical relationship is that equal stimulus ratios produce equal subjective ratios. This principle is the essence of the psychophysical "power law" governing human impressions of most physical sensations and is probably the most strongly supported law of human judgments in psychology.

Early in the development of magnitude scaling, it was found that the empirically obtained mathematical function relating numeric estimation to each sensory modality varied reliably between sensations; specifically, different sensations were found to grow at different rates. These results, however, were challenged by critics who argued that the sole reliance on numeric estimation made verification of the power law impossible independent of numbers. Consequently, the following technique was used. Rather than match numbers to stimulus intensities, subjects would, for example, use force of hand grip to respond to the brightness of light. A basic conclusion was that the power law is not dependent on numeric estimation, it also occurs with other response modalities. This also cleared the way for cross-modality matching in which quantitative response modalities, each of which grows at a known characteristic rate, are matched to each stimulus. An example would be responding to the stimulus of light with both loudness of voice and force of hand grip. Within the cross-modality paradigm, the use of two responses allowed validation of the magnitude scale of impression.

Cross-modality matching allowed those who believed in the techniques of psychophysics to extend it to the magnitude scaling of social-psychological impressions by the simple substitution of social for physical stimuli. Usually, words or phrases denoting instances (items) on a social-psychological dimension take the place of the physical stimuli traditionally used in classic psychophysics. The reasoning behind this was straightforward; some researchers came to believe that estimates of the intensity of physical stimuli—impressions of the brightness of light, loudness of sound, heaviness of lifted weight—are indeed judgments, in part as a consequence of successful applications of the cross-modality matching paradigm to social stimuli. Several criterion tests (see Lodge 1981) were then developed to validate a magnitude scale of social judgments. If these tests are satisfied, then the derived scale is labeled a "psychophysically validated ratio scale."

This procedure was applied to social science data to surmount the difficulties involved with ordinal data. As such, it was a worthy effort. One criticism is an almost compulsive obsession with finding linear examples of regression. Linear relationships are not necessarily the ultimate in regression analysis (see *chapter 6*); for example, the data might be described better by a nonlinear function. But, the need was appreciated for "something" better than forcing ordinal data into interval statistical techniques for analysis of social science data.

4.6.2 Studies of Visual Quality

How are signal detection theory, psychophysics, and magnitude scaling used in the landscape literature? In terms of theory, they support the claim that a comprehensive stimulus-response function describing landscape preferences may exist. More specifically, the goal of this body of literature is to ". . . explore the possibility of the existence of a standard psychophysical or stimulus-response function that specifies *a priori* the shape and character of the relationship between preference and dimensions of the landscape" (Buhyoff and Wellman 1980, p. 259).

In terms of methods, however, this same body of literature should be examined critically. Three different landscape preference studies conducted over a 4-year period on a wide variety of subjects used paired comparisons of landscape slides (Buhyoff and others 1978, Buhyoff and Leuschner 1978, Buhyoff and Reiseman 1979). Paired comparisons produce ordinal measurements; however, in all three studies the results were assigned interval scaling scores by invoking Thurstone's law of comparative judgment. Complicated regression techniques were then used to search for a stimulus-response function to describe the data.

Invoking Thurstone's Law is equivalent to the need for belief in psychological theory to have confidence in results of classical parametric tests on ordinal data (see *section 3.2*). Those working in the psychophysical tradition of landscape preferences seem to have missed the point of signal detection theory, psychophysics, and magnitude scaling. The trend has been to use the psychophysical approach to justify the continued use of parametric procedures such as regression on ordinal variables. Why? Partly because of tradition, partly because of the perceived need to make longitudinal studies within some fields comparable, and partly because ordinal scaling is less expensive and less time consuming than magnitude-type scaling alternatives.

When you read studies that involve an attitude variable, ask these questions:

- What method (e.g., Likert scaling, paired comparisons, etc.) has been used in the study to scale the attitude variable?
- Does the author of the research report claim that this variable has an interval level of measurement?
- If so, what psychological assumptions does the author make to try to rationalize this claim? Does the author invoke Thurstone's law of comparative judgment, for example?

5. VERIFY ASSUMPTIONS OF CORRELATIONAL ANALYSES

When the methods component of a study includes statistical considerations, it will have a given purpose in terms of examining variables. Of the four purposes listed in *chapter 2*, one was that of exploring relationships among variables. Several common statistical methods are used for this purpose in evaluating landscape quality. The most important conceptually are *correlation*, *factor analysis*, and *multidimensional scaling*. These techniques seem complex, but they are useful tools that help to explain what is happening in a data set with a large number of variables.

5.1 Definitions

5.1.1 Simple Correlation

Correlation usually denotes the degree of strength of relationship between random variables taken two at a time. Even though a study involves a large number of variables, examining the correlation between each possible pair of variables is usually instructive. Correlation is particularly useful in exploratory studies: if the strength of the relationship is high (i.e., close to 1 or close to -1), then for example we might be interested in trying to predict one variable from the other. Several measures of correlation for two variables are available. Examples are Pearson's product-moment correlation, Spearman's rank correlation, Kendall's tau, Phi-correlation, and intraclass correlation. The following chart summarizes the levels of measurement presupposed by these correlation measures:

Level of measurement of the first variable / Level of measurement of the second variable	Nominal	Ordinal	Interval
Nominal	Phi-correlation (two binary variables)		Intraclass correlation
Ordinal		Spearman's rank correlation; Kendall's tau	
Interval	Intraclass correlation		Pearson's product moment correlation

5.1.2 Factor Analysis

Factor analysis refers to a family of statistical techniques whose common objective is representing a set of variables in terms of a smaller set of hypothetical variables. It is based on the assumption that the smaller number of underlying factors are responsible for correlation among the variables. The simplest case is one in which one underlying common factor is responsible for the correlation between two observed variables.

For example, suppose 100 individuals are randomly selected from the population and their weight, height, blood pressure, etc., are measured. The measurements constitute observed variables and are interval-level measurements (see *section 3.1*). These basic data are then arranged systematically, in what is usually called a data matrix:

Entity	Variables 1	2	3 . . . 80
1	125	63	7
2	149	59	8
3	220	61	12
4	190	62	7
.			
.			
.			
100			

The data matrix has two dimensions. One is called the entity mode, which represents the cases—persons in this example—arranged as rows. The other dimension is the variable mode, which displays the observed measurements in columns. If 80 variables were measured for each of the 100 individuals and each measurement produced one value for a variable, then the data matrix would contain 100 people times 80 measurements or 8,000 numbers. The matrix would need to be simplified; it would contain too many numbers for easy comprehension.

The first step toward simplification is examining relationships among the variables. This can be done by forming a correlation matrix, in which the variables from the data matrix are both the rows and the columns. The values in the correlation matrix measure the association between each variable and each of the other variables (see *section 5.1.1*). A correlation matrix shows whether there are positive relationships among these variables (correlation values are greater than zero), negative relationships (correlation values are less than zero), and whether the relationships within some subsets of variables is stronger (correlation values are closer to 1 or -1) than that between the subsets.

Factor analysis is then used to address the question of whether these observed correlations can be explained by a small number of hypothetical variables, e.g., perhaps weight and height together tap a dimension having to do with body build. If the researcher has little idea as to how many underlying dimensions exist, factor analysis will uncover the minimum number of hypothetical factors that can account for the observed pattern of correlation, and allows an exploration of the data so that it can be reduced in size and analyzed economically. This is exploratory factor analysis. The majority of the applications in social science belong to this category. However, there is also confirmatory factor analysis. If the researcher, for example, believes at the start that different dimensions underlie the variables and that certain variables belong to one dimension or another, factor analysis confirms or tests these hypotheses. The division between the two uses is not always clear, and factor analysis has many methods and variants.

Factor analysis is virtually impossible to do without computer assistance. The usual approach is to input the correlation matrix into a factor analysis program and choose one of the many methods of obtaining the solution. (Several major alternatives are given in the literature, but the specifics can be safely ignored at this point.) The researcher specifies the number of common factors to be extracted or the criterion by which such a number can be determined.

Roughly, the program searches for a linear combination of variables (a factor) that accounts for more of the variation in the data as a whole than does any other linear combination of variables. The first factor is the single best summary of linear relationships exhibited in the data. The second factor is the second best linear combination of variables, given that it is not correlated with the first. That is, the second factor accounts for a proportion of the variance not accounted for by the first factor. Subsequent factors are defined similarly until all the variance in the data is exhausted.

The resulting set of factors is called the initial solution. A terminal solution—obtained by a complex procedure called rotation—may be a further simplification of the data, but it is beyond the scope of this report.

In general, in the fields of psychology and education, the main motivation behind use of factor analysis is finding the factor structure among a set of variables. This is not the motivation in other disciplines. Most other social science disciplines use factor analysis to simplify data by obtaining factor scales that can be used as variables in a different study. Factor scales are commonly analyzed along with other variables.

5.1.3 Multidimensional Scaling (MDS)

Multidimensional scaling is a set of mathematical techniques that enable discovery of the "hidden structure" of a data set. MDS operates on numbers that indicate how similar or different two objects are or are perceived to be. Such numbers are called *proximities*. Proximities can be ordinal-level measures. A correlation coefficient may be used as a measure of proximity. Proximities may be obtained during the data collection phases of a study by asking people to judge the similarity of a set of stimuli (e.g., photographs of landscapes). MDS results are displayed as a geometric pattern of points, like a map, with each point corresponding to one of the stimuli. This pattern is considered to be the hidden metric structure of the data. The greater the similarity between two objects, as measured by their proximity values, the closer they should be on the map. Generally, the most useful insights from MDS are gained by visually examining and interpreting the configuration. More complicated nonvisual techniques may also be used.

For example, suppose that each of 20 campers rated the degree of overall similarity between 10 landscape photographs on a scale ranging from 1 for "very different" to 9 for "very similar." No information would be given to the campers concerning the characteristics on which similarity was to be judged because the goal is to discover such information and not impose it. These data are input to an MDS computer

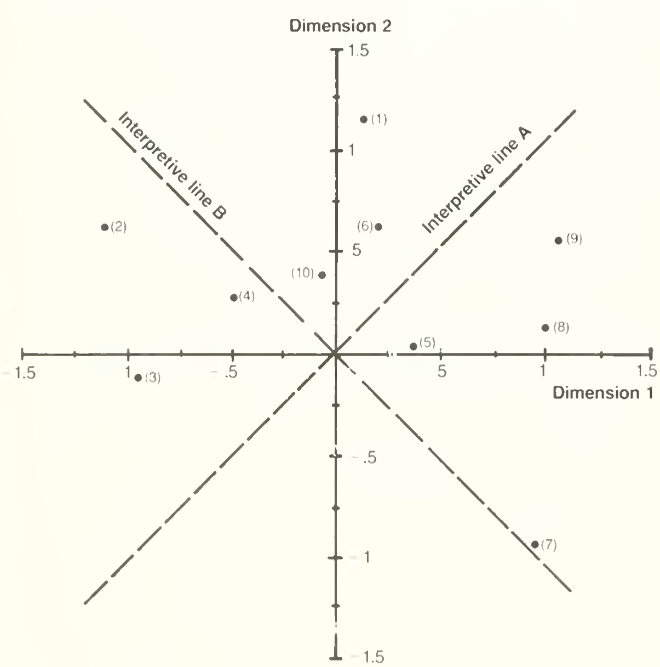
program. MDS calculations are complex, and even the simplest versions are always performed with the aid of a computer. Also, a large variety of different computational methods are used, some of which depend only on the rank order of the proximities.

The chief output of the MDS computer program is the map-like representation of the proximities data. The motivating concept is that the distance between the points should correspond to the proximities. In this sense, the output of MDS is not much different than a scatter diagram. MDS, however, can be mathematically complex. It is possible not only in two and three dimensions but also in four or more dimensions. This is impossible to visualize, but it can be dealt with mathematically.

Suppose, for the sake of simplicity, that the researcher starts with the results obtained from a two-dimensional analysis of these data. The computer output will include a list of coordinates for the landscapes and a plot of these values:

Stimuli (landscape)	Dimension 1	Dimension 2
1	0.15	1.12
2	-1.12	0.68
3	-0.90	-0.19
4	-0.50	0.29
5	0.36	0.02
6	0.19	0.64
7	0.96	-0.90
8	1.04	0.12
9	1.14	0.59
10	0.03	0.36

The most common way of interpreting such an MDS plot is looking for lines—possibly at right angles to each other—that would divide the data in some easily describable way. Interpretive divisions could be represented as dashed lines on the plot: Numbers in parentheses indicate which stimulus is plotted at that point.



Also suppose that the researcher knows something about these landscape photographs and might be able to interpret the results in a rough-and-ready fashion. For example, everything below line A is an example of a forested landscape. Everything above line A is an example of a nonforested landscape. Line B distinguishes both the forested and nonforested photographs in terms of how skilled the photographer was on the day(s) the photos were taken.

Researchers who use MDS virtually never stop at this simple level of analysis. Configurations are rotated (as in factor analysis), other statistical techniques may be applied for dimensional interpretation, etc. Analysis can get very complex indeed.

When reading a research report based on MDS, be aware that—because the techniques are relatively new and computer driven—users sometimes understand them less than they do other statistical procedures. Therefore, you should ask a statistician for help.

5.2 Assumptions and Pitfalls

The central aim of factor analysis and MDS is simplifying data without losing much information. Other factors aside, reliability of results can be judged by the degree to which the assumptions necessary for these techniques have been met.

Two basic assumptions underlie factor analysis. The first is that the observed variables are linear combinations of some underlying causal variables (factors). This assumption has to be substantiated by having some knowledge of the data and research problem to begin with. The second assumption involves the notion of parsimony: if both one-factor and two-factor models explain the data equally well, the convention is to accept the one-factor model. If either assumption is invalid, results can be fallacious or indeterminant. Factor analysis also requires interval level measurement of variables. When used with ordinal variables, several operations are not well defined in a statistical sense. The conservative approach dictates use of factor analysis on ordinal data for exploratory uses only and not for statistical inference.

The assumptions for MDS are less stringent. Interval level measurement is not required for some types of MDS. The only assumption required is that the subjects ranking the stimuli (photographs in the example) according to degree of similarity or difference know something about the items being ranked. This technique also does not require that the data have a multivariate normal distribution as does factor analysis. Therefore, even if it is little understood by most users, MDS is useful for describing the attitudes, opinions, or perceptions of the individuals doing the ranking. MDS can do what it is designed to do—including inferring a metric structure from nonmetric ordinal data. MDS, as developed by Roger Shepard (1962, 1963) at Bell Laboratories in the early 1960's, was partially a reaction against "Thurstonian" scaling procedures when used with psychological data.

The following tabulation summarizes the levels of measurement presupposed by factor analysis and multidimen-

sional scaling, according to whether these techniques are used for exploratory uses or for hypothesis testing.

Use	Level of measurement: Ordinal	Level of measurement: Interval
Exploring relationships	Factor analysis or MDS may be used	Factor analysis or MDS may be used
Testing hypotheses	Some types of MDS *only* may be used	Factor analysis or MDS may be used

- Is Spearman's rank correlation or Kendall's tau used to measure correlation between two ordinal-level variables? If so, the results are valid.
- Is Pearson's product-moment correlation used to measure correlation between two ordinal-level variables? If so, the results of the analysis are neither empirically nor semantically meaningful.

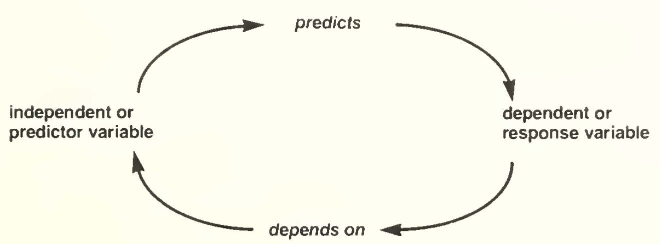
For the purpose of hypothesis testing (as opposed to exploratory use) . . .

- Is an appropriate type of multidimensional scaling used on ordinal-level variables? If so, the results are valid.
- Is factor analysis used on ordinal-level variables for the purpose of hypothesis testing? If so, the results of the analysis are neither empirically nor semantically meaningful.

6. EVALUATE SUITABILITY OF PREDICTION MODELS

6.1 Simple Models

The simplest kind of prediction model involves two variables. One is called an *independent* or *predictor variable*, the other variable is called a *dependent* or *response variable.*

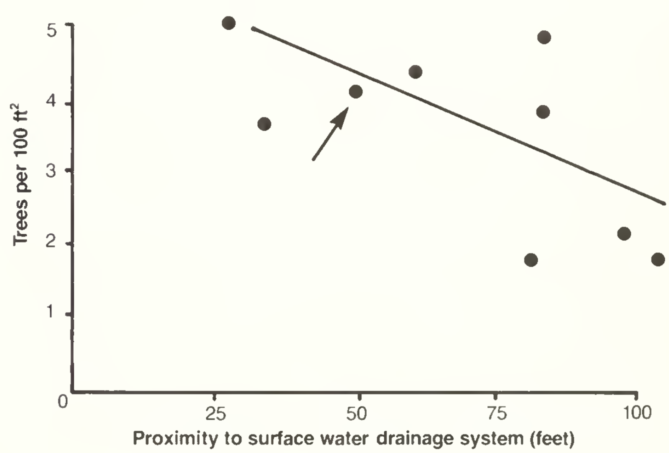


For example, a model might involve one variable measuring proximity to surface-water drainage system and a second variable measuring trees per 100 square feet. Proximity to surface-water drainage system is the independent variable, because the interest is in studying how and how well it may be used to "predict" the other variable. The dependent variable is therefore the number of trees per 100 square feet. Put another way, the object is to see how the number of trees per 100 square feet (the dependent variable) "depends on" proximity to surface-water drainage system (the independent variable).

6.1.1 Linear Models

Two variables can be related in literally hundreds of thousands of ways. One of the simplest is a *linear relationship*, meaning that it may be described by a straight line. For example, for every 25 yards closer to surface-water drainage system, the number of trees per 100 square feet will increase by one. Few real-world relationships are this simple. Sometimes, however, a straight line provides an approximation to a real-world relationship that is "good enough" to be useful.

The graph below is an example of data that are "approximately" linearly related:



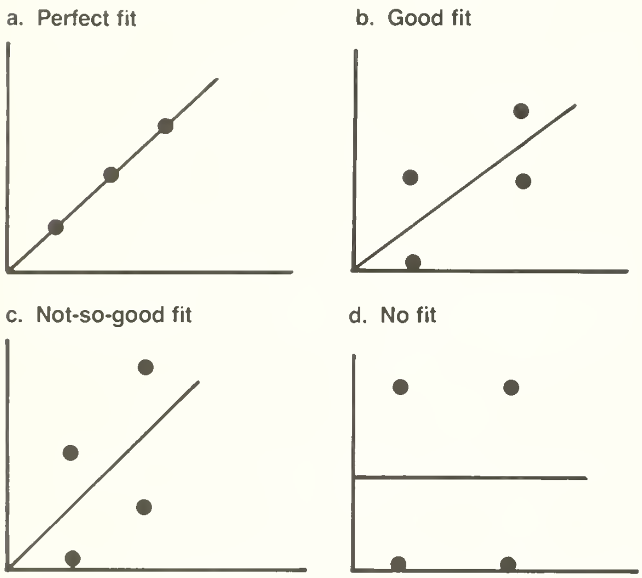
Proximity to surface water drainage system is plotted horizontally; number of trees per 100 square feet is plotted vertically. The sample consists of 9 pairs of observations. Each pair constitutes one sample point and consists of a measurement of each of the two variables. For example, 50 feet from surface water drainage system were four trees per 100 square feet (this point is marked with an arrow on the graph). The straight line in the plot is an "approximation" to the real data.

To describe the relationship between the two variables, the researcher no longer has to look at a list of data points. Such a list gets harder to "read" as the number of data points gets larger. Instead, a simple linear mathematical equation concisely summarizes the list of data points.

A central problem in using a straight line to approximate real data is finding the line that gives the best fit to the data. There are an infinite number of lines to choose from. The researcher will want to choose the straight line that is in some way "closest" to all the data points. Luckily, statisticians agree on the best method to use to choose the line: *least-squares estimation.* It is used universally in the computer programs

that researchers invoke to statistically fit straight lines to their data.

The procedure that produces the "best" straight line for a given set of data does not guarantee that a straight line will adequately describe the data. The graphs below illustrate lines that have been fit to data by least squares estimation.



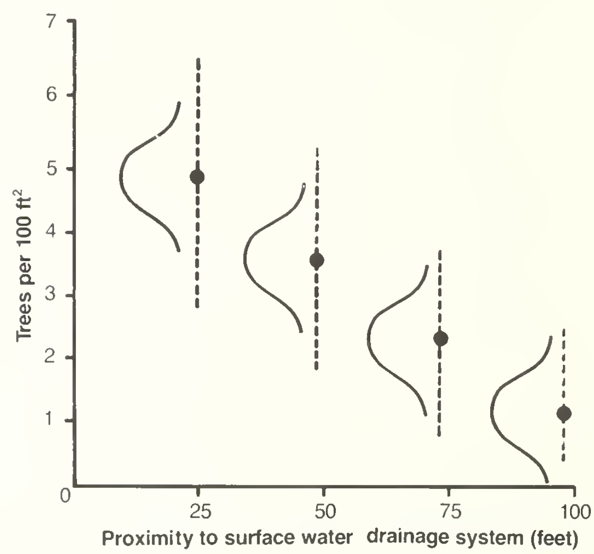
This "best" line is next to useless in summarizing the observed relationship between two variables in graphs c and d, because the data just are not linear!

I want to emphasize the difference between *model-building* and hypothesis testing (or inference). In model-building, the object is to describe a set of data points by a mathematical equation that provides a good approximate fit to these points. No assumptions are necessary because the researcher isn't saying anything about probabilities. In hypothesis testing, various assumptions are required because the researcher needs to know the probability of reaching a wrong conclusion. Five assumptions are necessary for inference with straight lines involving two variables. These assumptions involve how and where the data were collected.

Assumption 1. The values of the independent variable are interval level and are *fixed,* not random. That is, the researcher decides in advance of data collection upon the specific levels of the independent variable at which to take measurements on the dependent variable. This assumption would be satisfied for the example if—before collecting data—the researcher decided to measure number of trees per 100 square feet at each of the following distances from surface water drainage systems: 25, 50, 75, and 100 feet. The levels of the independent variable thus would be fixed at 25, 50, 75, and 100 feet. No data would be collected for other values of the independent variable.

Assumption 2. The values of the dependent variable for each level of the independent variable are interval level and are statistically independent random variables. If the researcher fixes levels of the independent variable, then the dependent variable will take on values at these levels with associated probabilities. That is, the values of the dependent variable for each level of the independent variable will be *random variables.* To test statistical hypotheses, the researcher must also assume that these values are statistically independent. Roughly speaking, statistical independence means that the occurrence of one observation in no way influences the occurrence of the others. Random sampling helps to assure the statistical independence of the measurements *(chapter 7).*

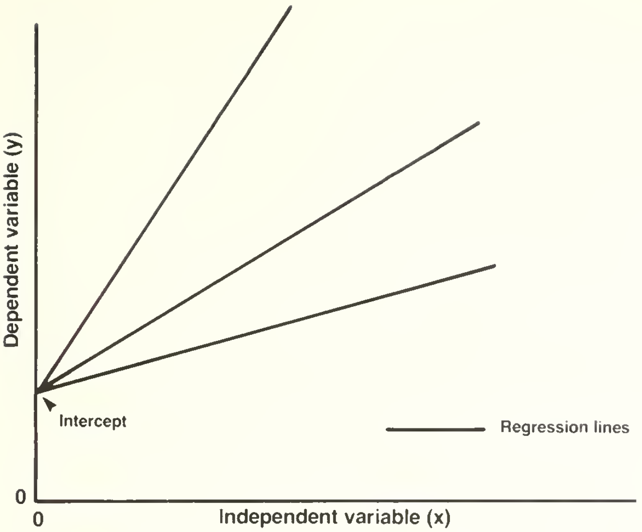
Assumption 3. For each value of the independent variable, the distribution of the dependent variable is normal. In the example, suppose the researcher works with four surface water drainage systems and therefore ends up with four values of the dependent variable at each level of the independent variable. Each set of values at each fixed level of the independent variable represent a set of observations sampled from a population consisting of many values. The assumption is that the population values have a normal distribution.



Assumption 4. The population variance of the dependent variables is the same for all values of the independent variable. Thus the normal distributions may all be situated in different places (have different means), but all have the same shape (have the same variance). This assumption and assumption 3 can be checked by past experience, or by present data if numbers of observations at each level of the independent variable are sufficient. Often, however, insufficient numbers of observations are collected, or else the researcher neglects to check.

Assumption 5. The regression is linear. That is, the means of the normal distributions described above lie over a straight line. These means are indicated by the points in the above graph.

The mathematical equation for a straight line involves two *parameters* (i.e., constants). These control where the straight line falls on the graph. One parameter is the intercept. It designates the point at which the line hits the vertical axis. The other parameter is the slope. It designates the change in the dependent variable that is associated with one unit of change in the independent variable:



The three regression lines in the above graph have the same intercept, but have different slopes (directions).

Several statistical hypotheses can be tested if the five assumptions above are met. These hypotheses include the following:

- The slope of the *regression line* is zero, indicating that the independent variable is not linearly related to the dependent variable.
- The slopes of two lines calculated from samples drawn from two independent populations are equal.
- The intercept of the regression line is equal to some specified value.

Often researchers will fit a regression line to data and proceed to test statistical hypotheses where one or more of the five assumptions are violated. What is the result?

The effect of departure from a specific assumption must usually be studied by analyzing specific cases. General conclusions of effects of departure from an assumption are therefore not rigorously derived mathematical results. Nevertheless, some general principles can be stated:

- The assumption of independence of the observations is vital to the accuracy of the analysis. Therefore, the researcher should strive to assure that the observations are independent.
- For modest sample sizes, moderate deviations from the assumption of normality have little effect on the accuracy of the analysis.
- Deviations from the assumption of equal variances have little effect on the accuracy of the analysis, if the numbers of observations occurring at each level of the independent variable are equal. Therefore, a researcher should strive for a research design with equal sample sizes for each value of the independent variable.

6.1.2 Log-Linear Models

Log-linear models are similar to linear models. A basic difference is that log-linear models are designed to operate on nominally scaled data (see *chapter 3*), whereas linear models are designed to operate on interval-level data. Data for a log-linear model are best viewed as arising from a table with several dimensions, one dimension corresponding to each variable. For example, a two-dimensional data table might look like this (one variable is sex, another is race):

	Race	
Sex	White	Non-White
Male	36	22
Female	54	23

Although log-linear models were originally formulated to operate on nominal data, recent focus is on developing models tailored specifically for use with *singly ordered tables*, i.e., one ordinal variable and *doubly ordered tables*, i.e., two ordinal variables. A two-dimensional singly ordered table might involve the two variables sex (nominal) and social class (ordinal). Categories of social class might be lower, middle, and upper. An example of a two-dimensional doubly ordered table might involve the two ordinal variables social class and preference. Categories of preference might be weak, medium, and strong.

The general log-linear approach involves an *a priori* assignment of scores to the categories of the ordinal variables. For the example of preference, scores might be 1—low preference, 2—medium preference, and 3—strong preference. The model could be specified with either equal or different (sometimes arbitrary) spacing between categories of the ordinal variable. Spacing parameters could also be estimated from the data by established statistical methods, which would be optimal from the standpoint of not requiring strong spacing assumptions.

The study of landscape quality and preferences often includes assessing whether intervention into a landscape (e.g., constructing a road) will affect public preferences for the landscape. Suppose the interest is not only in constructing a road, but also in choosing the type of road to construct: gravel, black-top, or cement. The intervention variable would have these three categories plus a fourth corresponding to no road. A log-linear model could be constructed for a singly-ordered table, in which one of the variables (type of road) is nominally scaled and the other (preference for the landscape) is ordinally scaled. Separate log-linear models could be applied to the same sample of individuals for different landscapes. Then, however, the results of the different models would have to be pieced together, which—as with general linear models—has not been systematized. That is, the manner in which results are pieced together remains arbitrary.

6.2 Complex Models

More complicated types of prediction models are possible by increasing the number of variables, or specifying a more complex relationship among the variables, or both. Suppose that instead of one independent variable the analysis includes two or more independent variables in conjunction with a

single dependent variable. Suppose the researcher wants to describe the relationship between the independent variables and the dependent variable as linear. Again, the method of least squares estimation can be used to find a straight line that best fits the data. Regression with two or more independent variables is called *multiple regression* (as opposed to simple regression). Multiple linear regression is basically similar to simple linear regression except for the complexity of the calculations. The same assumptions are necessary for inference.

For example, if data were collected on 20 independent variables, not all of them would be used in the final model, because the simplest model possible is desired to predict the dependent variable. Furthermore, a mathematical relationship other than a strictly linear one might be used. Whenever data on a set of independent variables is collected, two questions arise: (1) Which of the variables should be included in the regression model? (2) What mathematical relationship should be used to describe them?

Finding a subset of independent variables for the final model and a function between them that adequately predicts the dependent variable, is called model-building. The process destroys the inferential capabilities of the standard linear model. The reason for model building is not that a true model exists and just needs to be found. Instead, it is to predict approximately the dependent variable and to understand the phenomenon being studied. A simple model is needed and one that cannot be significantly improved.

First consider what mathematical relationship should be used to describe the relationship between the variables. Looking at the mathematical form of a simple line and two variables helps in exploring this question. Let Y denote the dependent variable, x denote the independent variable, a denote the intercept parameter, and b_1 denote the slope parameter. The linear equation is

$$Y = a + b_1 x$$

The mathematical form of the relationship between the independent variable and the dependent variable can be changed by "transforming" terms in the equation. Some common transformations include taking the natural logarithm of one or more terms, taking the square of one or more terms, or raising one or more terms to some other power. One of the reasons for *transformations* is improving the fit of the model to the data.

Hull and Buhyoff (1983) fit six different mathematical functions to an independent variable measuring distance to a topographic feature and a dependent variable measuring scenic beauty. These included a variety of transformations:

$$\ln(Y) = \ln(a) + b_1 \ln(x)$$
$$Y = a + b_1(x^2)$$
$$Y = a + b_1 \ln(x)$$
$$\ln(Y) = \ln(a) + b_1 x$$
$$Y = a + b_1 x + b_2 x^2$$
$$Y = a + b_1 x$$

The dependent variable (scenic beauty) in this study was an ordinally scaled variable (see chapter 3) but it was treated as an intervally scaled variable and several statistical tests were performed for each of the six functions. While their model-building exercise was appropriate, the results of their hypothesis tests are suspect because ordinality has destroyed the inferential capabilities of the model.

Now consider which variables to include in the model. Virtually all regression analyses involving more than two variables are done by computer. A procedure that helps to select best candidates of variables and transformed variables is called *stepwise regression.* There is no unique best subset of variables. One reason to use transformations is to improve the fit of the model. Another reason is that transforming one variable will sometimes result in a simpler model—one with fewer terms.

Suppose a certain model (variables plus functional relationship between them) is appropriate to study certain phenomena. This appropriateness can be established by building the model with one sample of data and testing the fit of the model on a separate sample of data. Then the model can be used to predict the dependent variable from the independent variable. For example, suppose that in the study of the effect of proximity of surface-water drainage system (x) on number of trees per 100 square feet (Y), the researcher found the following functional relationship:

$$Y = 6 + (-.04)x$$

Then for x = 90, Y = 6 - .04(90) = 2.4. In other words, at 90 feet from the water drainage system, the researcher predicts 2.4 trees per 100 square feet.

Models with many independent variables give more information about potential causal factors for the phenomena being studied. While including variables in a regression equation does not imply causality, the inclusion of certain factors over others in the equation helps aid understanding of the mechanisms at work in the phenomena.

If the primary statistical purpose of the study is building prediction models using variables ask the following questions:

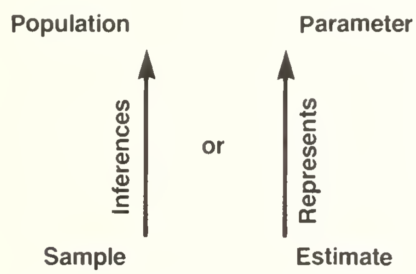
- Which variables are dependent (i.e., response) variables?
- Which variables are independent (i.e., predictor) variables?

If the researcher used simple linear regression (or multiple regression) and tested hypotheses, were the following assumptions met:

1. Values of the independent variables are interval level and fixed.
2. Values of the dependent variable for each level of the independent variable are interval level and are statistically independent random variables.
3. For each value of the independent variable, the distribution of values for the dependent variable is normal.
4. Population variance of values for the dependent variable is the same for all levels of the independent variable.
5. Regression is linear.

7. DETERMINE REPRESENTATIVENESS OF SAMPLE

Researchers usually want to generalize about a whole class or *population* of individuals or things; however, for reasons of cost, time, and practicality, this is not really possible. Only part of a population can be examined, and this part is called the *sample.* A researcher then makes generalizations from the part to the whole, or more technically a researcher makes inferences from the sample to the population:



What a researcher usually wants to estimate are certain numerical facts or parameters about the population of interest. An example is how many pines per acre grow in a National Forest. The forest is the population, and the number of pines per acre is the parameter. Because limited time and money prohibit counting all of the pines, the researcher samples a few acres of the forest, counts the pines on those acres, then estimates the number per acre in the population. Because parameters like this one cannot be determined with total precision, a major issue is accuracy—how close is the estimate from the sample to the actual parameter in the population?

Parameters are estimated by statistics—the numbers that can be computed from a sample. Statistics are known, parameters are unknown. Estimating the parameters of a population from a sample is justifiable if the sample represents the population. In general, (1) the method of choosing the sample determines whether the sample is representative; and (2) the best methods of choosing a sample involve the planned introduction of chance or probability. The main reason for probability samples is to avoid *bias,* which can be defined as systematic error. Many different types of bias exist. A few of these are discussed in *section 7.3.*

7.1 Nonrepresentative Samples

The estimate of a parameter will be fairly accurate if the sample is obtained by chance. If the sample is obtained by human judgment then the estimate will be biased, that is, it will systematically deviate from the true population parameter. The reason is that human judgment is not impartial, but chance or probabilistic methods are impartial.

Convenience sampling and *quota sampling* are methods that involve human judgment. In convenience sampling, the units sampled are those readily available. An example would be choosing acres that are near roads in the National Forest, then counting the pines in those acres. Such a sample would not represent the population at large and would be biased, unless the researcher is lucky.

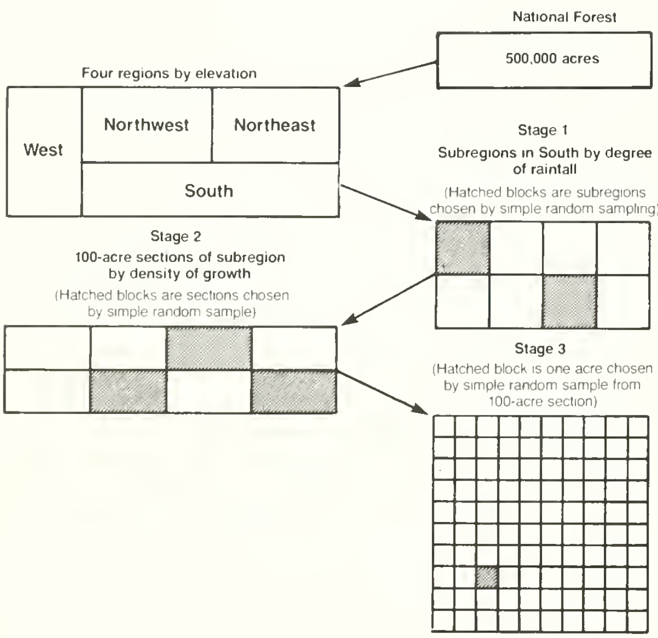
In quota sampling the National Forest would be canvassed. The goal is to find a perfect cross section of the forest on all the key variables that relate, say, to the existence of pine trees. Each person would be assigned a fixed number of acres to count pines, a fixed number of acres having pines of certain types, heights, etc. This type of sampling involves several difficulties. First, assigning quotas of type, height, or whatever about pines involves circular reasoning; the quotas are what you are trying to find out about the population (the parameter) and are not information that can be taken from other sources and used to construct a sampling procedure. Second, within the assigned quotas of acres, assistants are free to pick any acre they like. This leaves a lot of room for human choice, which is always subject to bias. In short, in quota sampling the sample is handpicked to resemble the population with respect to some key characteristics. This method seems sensible, but does not work well in practice. It was, for example, the sampling scheme used by Gallup, Roper, and others, that predicted Dewey would be elected president in 1948; Truman won the election. The interviewers used their own discretion to fill their quotas and chose too many Republicans, because they are marginally easier to interview in terms of having phones, living in nicer neighborhoods, or having permanent addresses.

7.2 Probability Samples

How is chance used to draw a sample? Assume that the National Forest in the example is well plotted in terms of acres and their boundaries, and that the forest is composed of 5,000 acres. Each acre is assigned a separate number, all 5,000 numbers are placed in a bin, and a statistician helps to decide that 200 of them should be drawn at random in order to have statistically significant results. Because there would be no point in counting the pines on the same acre more than once, the 200 draws are made without replacement. In other words, the bin is shaken well to mix up the plot numbers, and one is drawn out at random and set aside, leaving 4,999 in the bin. The bin is shaken again and the procedure repeated until 200 plot numbers have been chosen. These 200 plot numbers form the sample, called a *simple random sample.* The plot numbers simply have been drawn at random without replacement, and at each draw, every remaining plot number has an equal chance to be chosen. No human discretion is used, and the procedure is impartial—every acre has the same chance of getting into the sample.

For various reasons, simple random sampling often is impractical, and *multistage cluster sampling* is used instead. Suppose the National Forest is composed of 500,000 acres instead of 5,000 acres. Drawing plot numbers at random, in the statistical sense, becomes difficult. And because the forest is much larger, visiting the right acres becomes expensive.

In multistage cluster sampling, the forest would be separated into regions that are similar to each other. Suppose four regions are distinct in terms of elevation. Within each region, subregions could be grouped on the basis of another similarity, such as amount of rainfall. Then a simple random sample of these subregions would be taken. Only the selected subregions would be visited. This completes the first stage of sampling. Each subregion would then be divided into 100-acre sections on the basis of some other index of similarity—perhaps density of growth. At the second stage of sampling a simple random sample of these 100-acre sections would be drawn. At the third stage of sampling one acre would be drawn at random from the 100 acres in each selected section. The pines in this 1-acre plot would then be counted. This is a rather crude example of a somewhat complicated concept:



Multistage cluster sampling eliminates selection bias because it eliminates human choice. In summary, all probability methods for sampling have two critical properties in common with simple random sampling:

(1) A definite procedure exists for selecting the sample, and it involves the planned use of chance, and

(2) The procedure involves no human discretion as to who is interviewed or what part of the environment is sampled.

7.3 Possible Biases

In terms of sampling, selection bias is the most obvious type of bias. *Selection bias* is a systematic tendency on the part of the sampling procedure to exclude one kind of person or thing from the sample. For example, acres over a certain elevation might be excluded because of the difficulty of getting sampling personnel and equipment to the site. If a selection procedure is biased, taking a larger sample doesn't help, but just repeats the basic mistake on a larger scale.

A second common form of bias in terms of sampling is *nonresponse bias*. This occurs when a large number of those selected for a sample do not in fact respond to the questionnaire or interview. When counting pine trees this problem does not arise, but in other types of research it does. Nonresponders usually differ in terms of social class, education, values, etc., from responders. This means that the information obtained in the sample is distorted because it does not truly represent the whole population of interest. To compensate for nonresponse bias, a researcher can give more weight in the analysis to responses from people who were available but hard to get.

The basic formula presented so far concerning an attempt to choose a representative sample is

$$\text{estimate} = \text{parameter} + \text{bias}.$$

Any sample used to derive an estimate, however, is still only part of a population. Our estimate is likely to be a bit off in terms of accurately estimating the parameter. If a sample is part of a population that is chosen at random, then the amount by which an estimate misses a parameter is controlled by chance. A more accurate equation then is

$$\text{estimate} = \text{parameter} + \text{bias} + \text{chance error}.$$

Where the cost of a study fits into this equation is unclear. No straightforward relationship exists between cost and bias, for example. Depending on the type of study and the capability of the investigators involved, less expensive methods can be less biased than more expensive ones. Methods are often chosen by the researcher (not the statistician).

In general, examine any sampling procedure and try to decide if it is good or bad. Many, if not most, samples are unsatisfactory. If the sample is unsatisfactory, then the results of analysis must also be unsatisfactory. To decide if a sample is satisfactory, first examine how it was chosen. Were probabilistic methods used? If not, was there selection bias? Was there nonresponse bias? Are any other sources of bias obvious in the sampling procedure?

GLOSSARY

alternative hypothesis: the opposite of the null hypothesis; an educated guess that the distribution of the observed values does show a difference from the assumed probability distribution

association: two variables are highly associated if you can use the value of one variable to predict the value of the other with confidence

bias: any process at any stage of inference which tends to produce results or conclusions that differ systematically from the truth

convenience sample: the units sampled are those readily available (not a good sampling method)

correlation: a measure of association, usually between two random variables; correlation values close to 1 mean strong positive association; values close to -1 mean strong negative association; 0 means no association

dependent variable: a variable that "depends on" another variable, i.e., a variable that is predicted by another variable

descriptive statistics: techniques to organize and summarize data

doubly ordered table: a table with two ordinal variables

empirical distribution function (cumulative relative frequency): the proportion of responses up to and including a specified response category

factor analysis: a family of statistical techniques designed to represent a set of (interval-level) variables in terms of a smaller set of hypothetical variables

fixed values: values of the variable are fixed in advance of data collection

hypothesis test: a type of statistical inference that involves comparison of an observed value (or values) with a value (or values) derived from probability theory

independent variable: a variable used to predict another variable

interval and ratio scales: result from measuring things with a physical unit, such that the exact distance between things is established

least squares estimation: the method commonly used to choose the straight line that is closest to the data points

Likert Scale: uses multiple stimuli to produce a single attitude measure

linearity: (see linear relationship)

linear relationship: the relationship between a dependent variable and one or more independent variables may be described by a straight line

magnitude scaling: the attempt to measure interval-level preference by using psychophysical scaling methods

mean: arithmetic average

median: the value that divides the ordered observed values into two groups of equal size

mode: the value that occurs most often

model-building: finding a subset of variables and a function between them that adequately predicts a dependent variable(s)

multidimensional scaling: similar to factor analysis; "proximities" among objects are used as data

multiple regression: regression with two or more independent variables

multistage cluster sampling: a type of probability sample that is conceptually more complex than the simple random sample but on the other hand is more practical when the population is large

multivariate: involves two or more variables

nominal scale: results from sorting things into homogeneous categories

nonresponse bias: occurs when a large number of those selected for a sample do not respond to the questionnaire or interview

normal probability distribution: a symmetric distribution specified by two parameters and the following equation:

$$f(x) = \frac{1}{\sqrt{2\pi}\,\sigma} e^{-(x-\mu)^2/2\sigma^2}$$

The distribution is sometimes called an "error curve" or Gaussian distribution or "bell-shaped" curve. σ^2 is the population variance

null hypothesis: an educated guess that the distribution of the observed values basically does not differ from the assumed probability distribution

operational definition: definition of a concept that includes procedures for classifying and measuring the phenomenon

ordinal scale: results from sorting things into homogeneous categories that are ordered or ranked with respect to the degree, intensity, or amount of something they contain

paired comparisons: subjects are asked to judge pairs of stimuli

parameters: constants that determine the shape and location of a distribution; facts about the population of interest

parametric statistics: statistics that assume a parametric model, i.e., a model with finite numerical parameters or facts

percentile: the specific value of a random variable that corresponds to a given cumulative probability

population: a whole class of individuals or things

predictor variable: independent variable

proximity: a measure of how similar or how different two objects are

probability model: translation of an empirical problem into probabilistic terms

Q-sort: subjects are asked to sort stimuli into a fixed number of categories in terms of the degree to which each stimulus represents the concept

qualitative: extends the idea of physical measurement to include various categorization procedures

quantitative: involves physical measurement

quota sampling: assigning quotas to characteristics of the population and then sampling to achieve these quotas (not a good sampling method)

random variable: a variable that has probabilities associated with each possible numeric value

range: largest value in a set of observations minus the smallest value

rank-ordered scales: example is the method of paired comparisons (see above)

rating scales: example is the Likert Scale (see above)

ratio scales: (see interval and ratio scales)

raw data: the values collected for each variable, without any statistical or other manipulation having been done to alter or adjust them

regression line: a line fit to the data using least squares estimation and analyzed within the framework of statistical theory of regression

response variable: dependent variable

sample: part of the population

sample variance: the sum of squared deviations of each observation from the mean, divided by the number of observations minus 1

selection bias: a systematic tendency on the part of the sampling procedure to exclude one kind of person or thing from the sample

semantic differential: subjects are asked to decide to what degree a concept is associated with selected sets of bipolar adjective pairs

simple random sample: each sample unit has an equal chance of being chosen (best sampling method)

singly ordered table: a table with one ordinal variable

standard deviation: the square root of the sample variance

statistical hypothesis: a statement concerning the distribution of probabilities for different values of a random variable

statistical inference: statistical methods designed to assess the impact that chance variation has on research results

statistical independence: the value obtained for one observation does not affect the values we are likely to get for other observations

stepwise regression: a computerized procedure for selecting best candidates of predictor variables in a regression problem

theoretical definition: definition of a concept in terms of other concepts which supposedly are already understood

transformations: a function of a variable (any variable—dependent, independent, etc.—can be transformed)

univariate: involves one variable

variable: something that is measured or counted

Z-scores: normally distributed random variables that have been converted to units of standard deviations relative to the mean

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APPENDIX—ADDITIONAL READING

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Publication of a research report does not guarantee that its results and conclusions are statistically valid. Each statistical method serves a particular purpose and requires certain types of data. By using this report as a guide, readers of research reports can better judge whether the statistical methods were appropriate, how closely measurements represent the concept being studied, and how much confidence to place in the conclusions. Descriptions of sampling methods and of possible biases show how results can be better evaluated with respect to sampling.

Retrieval Terms: scaling of attitudes, statistical assumptions, ordinal data analysis, sampling biases

United States
Department of
Agriculture

Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-88

Developing Fire Management Mixes for Fire Program Planning

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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

May 1986

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CONTENTS

INTRODUCTION

Economic efficiency evaluation of fire management program options is vital in the long-term fire management planning process (Oregon State Dep. of Forestry 1972; Schweitzer and others 1982; U.S. Dep. Agric., Forest Serv. 1980, 1982; Winkworth and others 1981). And more and more, economic efficiency is being evaluated through the use of simulation models (Bratten and others 1981; Mills and Bratten 1982; Schweitzer and others 1982; U.S. Dep. Agric., Forest Serv. 1980, 1982), which permit evaluation of a broad range of fire management program options. Information required by such models includes the cost and benefits of all firefighting inputs used, resulting changes in net value of resources, and the combination of inputs that would be needed to execute the fire program being evaluated. The funding levels of fire programs are translated into a combination of firefighting inputs (for example, vehicles, aircraft, bulldozers, and crews) that would be used in the program. Such a combination of particular inputs in certain proportions is a *fire management mix*.

This report describes an algorithm by which the proportions in a fire management mix can be translated automatically into numbers of firefighting inputs available in a specific fire service area. The procedure was originally used in the prototype version of the Fire Economics Evaluation System (FEES), a simulation model being developed by the Forest Service to evaluate program options for nonsite-specific areas (Mills and Bratten 1982) (*appendix*). This report also describes how the concept of fire management mix can be applied to other fire management planning simulation models.

FINDING THE OPTIMUM PROGRAM LEVEL

Cost plus net value change (C + NVC) is the most widely accepted measurement for determining optimal fire management program expenditures (Gorte and Gorte 1979). Although the C + NVC concept has been modified over time, the fundamental concept is still the same: at the optimum program level, the sum of fire protection costs and the value of net change in resources is minimum. That is, the best program level will provide the most fire protection with minimal expenditures, and will result in the least net fire damage (damages minus benefits). Expenditures are combined costs of presuppression (prevention, detection, initial attack, and fuel management), and suppression. As presuppression expenditures increase, for example, net fire damages and suppression expenditures decrease.

Simulation models used in analyzing economic efficiency of fire program options evaluate C + NVC for different combinations of fire management mixes (FMM) and program levels (PL) (Bratten and others 1981; Mills and Bratten 1982; U.S. Dep. Agric., Forest Serv. 1985). A model starts by selecting an FMM. Development of the FMM requires two steps: (1) total program level dollars are allocated to fire program functional activities (prevention, detection, initial attack, and fuel management) through one set of weights (γ); and (2) functional activities dollars to be used during the fire season are allocated to specific firefighting inputs (FFI) through a second set of weights (α). The second set of weights represents the emphasis towards different kinds of FFI's, for example, heavy towards ground crews versus air attack. A different set of weights would emphasize different FFI's. Historical ranges of emphases in functional activity indicate ranges of FMM that can be used in evaluating fire program alternatives (*appendix*). The historical program level and distributions of program functional activities dollars should, however, serve only as a guide in the range of alternatives to be evaluated. They should not establish bounds for selection of analysis limits.

A model then simulates a fire season for the planning area selected and derives a C + NVC value for the FMM and PL. Each combination of PL and FMM determines one C + NVC point for a FMM (Mills and Bratten 1982). Theoretically, the curve is U-shaped with the minimum point representing expenditures of the optimum fire management program (*fig. 1*). To obtain other points on the same curve, the PL is varied while the FMM is held constant. The iterative process continues until a point is found at which C + NVC is minimum for the specific curve (local minimum—point A in *fig. 2*). To trace a full C + NVC curve, one of the two parameters involved in the analysis must remain constant. If the FMM and PL are changed at the same time, the same C + NVC curve is no longer being traced. However, the same point may be on two curves that intercept (point Y in *fig. 2*). Increasing PL alone can also trace more than one curve, if a higher PL would allow use of different, more costly firefighting inputs.

A model evaluates FMM's for other PL's until a global C + NVC minimum for the family of C + NVC curves is found (point B in *fig. 2*). The global minimum represents the optimum PL of all PL's and FMM's tested. Because all possible combinations of PL's and FMM's are not evaluated, the minimum

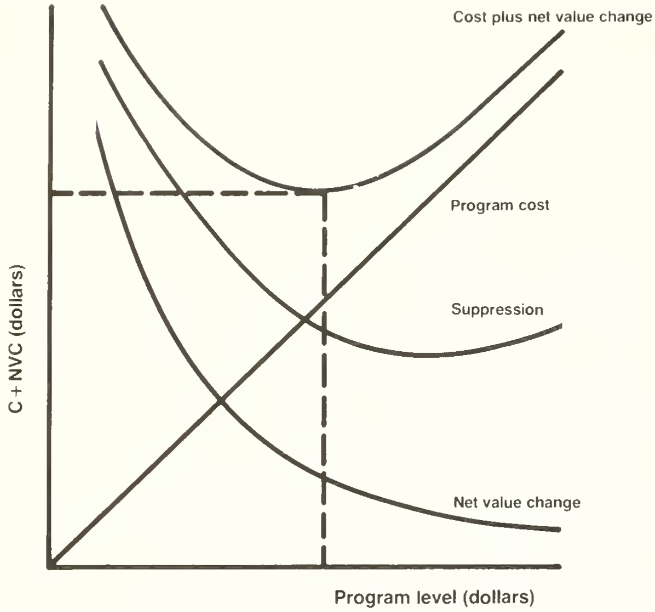

Figure 1—Minimization of the sum of cost plus net value change identifies the optimum fire management program level. The dashed lines indicate the cost plus net value change in dollars and the program level that correspond to the minimum cost plus net value change.

point selected may not necessarily coincide with the theoretical minimum.

The foregone present net worth of a fire program budget constraint can be derived from C + NVC curves. If the fire program level is constrained at PL_1 (*fig. 2*), the foregone present net worth of the fire program option is $(C+NVC)_1 - (C+NVC)_i$. Similar interpretation can be given to nonbudgetary restrictions that lead to the selection of a nonoptimum fire program.

Only initial attack and fuel management functional activities are usually evaluated with current simulation models for two reasons: (1) over 80 percent of total fire program dollars are allocated to initial attack and fuel management; and (2) deriving production functions for prevention and detection is difficult. The procedure described below is for only the initial attack function, although the same process can be used for developing input lists for all other functions.

The cost of the fire function usually covers the training and overhead expenses of nonfire funded personnel who occasionally do fire-related work (e.g., Forest Service regular employees organized in firefighting crews). This cost is charged to the fire function as the availability cost of these crews. Because the number of these crews is assumed to be constant throughout the range of program levels tested, their cost is subtracted from the initial attack program budget. The remaining budget is the amount available for buying other inputs for initial attack. Total dollars allocated to prevention, detection, and fuel management will remain constant for the example presented here so the program level increment goes to only the initial attack function. The procedure is flexible enough to permit increments in all functional areas.

TRANSLATING PROGRAM LEVEL INTO FIREFIGHTING INPUTS

The FMM controls the translation of total PL dollars into a firefighting input list that contains numbers of firefighting inputs. It can be obtained by using a seven-step algorithm (*fig. 3*) modified from the one developed by Hunter (1981).

Step 1: Allocate Program Level Dollars to Functional Activities

$$PL' = PL - \Phi - \Delta \quad (1)$$

$$F_i = PL\,(\gamma i) \quad (2)$$

$$PL' = \sum_i F_i \quad (3)$$

$$\sum_i \gamma_i = 1 \quad (4)$$

in which

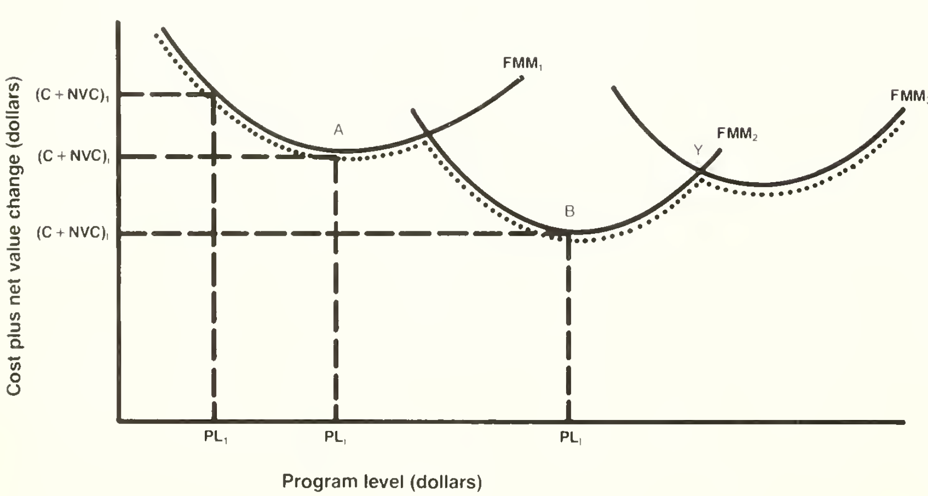

Figure 2—A family of C + NVC curves results from evaluation of several fire management mixes and program levels for a single fire service area. Each curve corresponds to a given fire management mix at a different program level. The dotted line indicates the frontier of greatest efficiency. Point A is a local minimum (for one curve), B is the global minimum (for the family of curves), and Y is a common point on two different curves that intercept.

PL = total fire program dollars for a particular fire service area

PL′ = fire program dollars allocated for the functional activity being evaluated in the model, for example, dollars allocated for initial attack

F_i = total dollars allocated to functional activity i (i = 1, 2, 3, . . . I)

γ_i = percentage of total program dollars allocated to functional activity i

Φ = total dollars of functional activities not evaluated in the model, for example, dollars allocated for prevention, detection, and fuel management

Δ = availability cost for nonfire funded personnel

Step 2: Allocate Functional Activity Dollars to Firefighting Inputs

$$B_{ij} = \alpha_{ij} F_i \qquad (5)$$

$$\sum_j B_{ij} = F_i = PL'\gamma_i \qquad (6)$$

$$\sum_i \sum_j B_{ij} = \sum_i \sum_j \alpha_{ij} F_i = PL' \qquad (7)$$

$$\sum_j \alpha_{ij} = 1 \qquad (8)$$

in which

B_{ij} = total dollars allocated to FFI_j

α_{ij} = percentage of F_i allocated to FFI_j (relative weight)

i = functional activity identifier (i = 1, 2, 3, . . . I)

j = fire management input identifier (j = 1, 2, 3, . . . J)

If only one functional activity was evaluated, i.e., i = 1, then equation 7 would be written as:

$$\sum_j B_{ij} = PL' = PL - \Phi - \Delta$$

Step 3: Translate Dollar Allocation into Physical Units

$$\Psi_{ij} = \frac{\alpha_{ij} F_i}{(C_{ij})(W_{ij})(S_{ij})} \cdot \frac{1}{\Omega_{ij}} \qquad (9)$$

in which

Ψ_{ij} = number of FFI_{ij} units in the input list

C_{ij} = unit cost per hour of FFI_{ij} (dollars)

W_{ij} = length of workday for FFI_{ij} (hours)

S_{ij} = length of fire season (days)

Ω_{ij} = fraction of total FFI_{ij} cost in the input list paid for by the particular fire service area

The staffing level of operations, the length of fire season, and level of FFI's available from cooperators must be taken into account to develop a realistic input list (IL). An IL is derived using a constant staffing assumption and a fixed fire season for the area evaluated. Constant staffing means that an input will be available for 8 hours of work daily during the fire season. In the planning phase, only the time necessary for the FFI regular tour of duty for the planning period is required.

In some parts of the country, such as Minnesota, in which the fire season is discontinuous, the constant staffing assumption may not be valid. The staffing level and organizational capabilities will most likely differ between the two segments of the season. But because agencies' budgets are allocated at the beginning of their operational year and spent as needed throughout the year, the proposed procedure can still be used. During the planning phase, a constant fire season is assumed and an input list is developed; the availability of the determined input list is then allocated according to the needs of the discontinuous fire season.

Because $\alpha_{ij} F_i = B_{ij}$ and letting $[(C_{ij})(W_{ij})(S_{ij})] = SC_{ij}$, then equation 9 becomes

$$\Psi_{ij} = \frac{B_{ij}}{SC_{ij}} \cdot \frac{1}{\Omega_{ij}} \qquad (10)$$

in which

SC_{ij} = seasonal cost of FFI_{ij}

In equation 10, when $\Omega_{ij} = 1$, the area under study is paying for the total cost of the FFI's on the list. The probability of getting the input when requested depends on the occurrence of multiple fires within the area. If $\Omega_{ij} < 1$, the area pays only for a fraction of the total cost of the input. In this case, the input list number

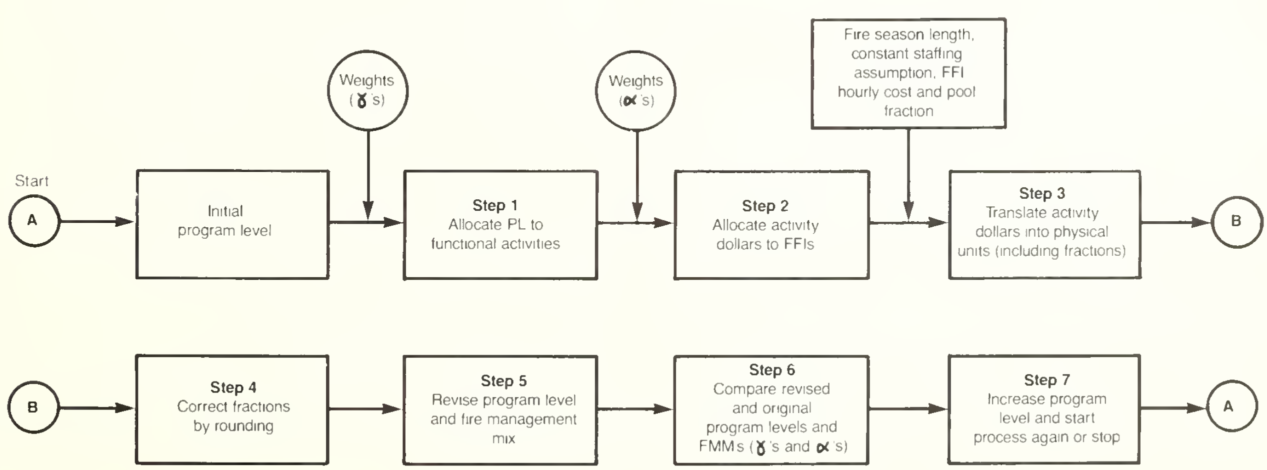

Figure 3—The firefighting input list for a given fire management mix and fire service area can be obtained automatically with a seven-step procedure.

(Ψ_{ij}) represents a pool of shared firefighting inputs from which a service area larger than the area under study is served. The probability of getting an input when requested is a function of multiple fire occurrence in the whole service area and not just in the area requesting the input. The initial attack section of most fire simulation models has a procedure to determine both the probability of multiple fire occurrence and the probability of getting the input requested from a shared pool. Thus far, only three firefighting inputs have been identified as shared between service areas: air tankers, smoke jumpers, and Category I crews.

Current simulation models deal very well with multiple fire occurrence situations between fire service areas, but not with multiple fire occurrence situations within service areas; that is, when one subsector of the service area is experiencing multiple fires while the rest of the service area is fire-free. In this case the problem is not one of availability of inputs, but one of time to attack. Most analysts believe that, by and large, the problem of multiple fires falls into this within-service-area category.

The result of the allocation process is the IL or inventory of all the FFI's available to control fires in a particular fire service area:

$$IL = \Psi_{11}, \Psi_{12}, \Psi_{13}, \ldots \Psi_{21}, \Psi_{22}, \Psi_{23}, \ldots \Psi_{NK} \qquad (11)$$

A perfect divisibility assumption does not necessarily hold in the specific case of firefighting inputs. The problem of lumpiness (noninteger solutions) occurs because fractional parts of air tankers, water pumpers, or tractors cannot be purchased. The solution obtained by using equations 1 to 11 does not guarantee an integer solution.

As shown in equation 10, solving for Ψ_{ij} will give the number of units of FFI_{ij} purchased with the total dollars allocated to FFI_{ij} (B_{ij}), the FFI_{ij} season cost (SC_{ij}), and the fraction of FFI_{ij} total cost paid by the particular service area (Ω_{ij}).

Let B_{ij} = \$225,000, SC_{ij} = 100,000, and Ω_{ij} = 1, and then solve equation 10:

$$\Psi_{ij} = \frac{B_{ij}}{SC_{ij}} \cdot \frac{1}{\Omega_{ij}} = \frac{225{,}000}{100{,}000} \cdot \frac{1}{1} = 2.25 \qquad (12)$$

Given the above conditions, 2.25 units of FFI_{ij} can be purchased. If input j were fire engines (with crews), for example, either 2 or 3 could be purchased.

Step 4: Correct Input Fractions by Rounding

To solve the lumpiness problem, a simple rounding procedure is recommended, i.e., whenever fractions occur in any one of the FFI's on the IL, they will be rounded to the next higher unit if the fraction is greater or equal to 0.5, and rounded to the next lower number if smaller than 0.5.

Step 5: Revise Program Level and Fire Management Mix

After the fractions have been corrected, the dollar amount in the functional activity being evaluated and the program level must be adjusted accordingly to account for the increase (or decrease) in costs and FFI for the fire season. The resulting adjusted input list is equal to

$$IL = U_{11}, U_{12}, U_{13}, \ldots U_{21}, U_{22}, U_{23}, \ldots U_{NK} \qquad (13)$$

As a result of the integer adjustment, the FMM also needs adjustment. Because of the integer rule, the dollar amount allocated to each functional activity being evaluated and the dollar amount allocated to each FFI within the functional activity may increase or decrease in relation to the original amount. If the amount does increase or decrease, then the proportion (γ_i) of the total dollars allocated to functional activities could change. The proportion (α_{ij}) of the total functional activity dollars allocated to each FFI_{ij} may also change.

Step 6: Compare Revised and Original Levels and Mixes

The new FMM and PL are compared with the original FMM and PL to determine if the same FMM is being measured. If the mean absolute value of the change in the α's from the original to the new FMM is 10 percent or less, the FMM is assumed to be the same. The FMM and PL selected are then evaluated in the simulation model to estimate one point on the C+NVC curve for the fire service area being evaluated.

Step 7: Increase Program Level and Repeat Procedure

The PL is increased and the process is repeated. Changes greater than 10 percent in the mean absolute value of the α's imply that a different FMM is evaluated. In that case, the PL would be increased by a selected amount, and the IL development process would be repeated until the mean absolute value change of the FMM is at or below the 10 percent critical value. If the critical value is not met within 0.90 of the selected increment, the evaluation process begins again at the next program level.

The iterative evaluation process continues until all selected PL and FMM are tested and an IL is derived for each. All PL, FMM, and IL meeting the critical value provide points to estimate a unique C+NVC change curve for a fire service area. The firefighting IL's not meeting the critical value criteria are also printed in a summary table (properly identified) of the program output so they can also be analyzed if managers believe the lists to be representative of the management emphasis identified, such as wilderness area or intensive timber, for the service area evaluated.

Different sets of weights (α_{ij}) can be used for different FMM in different kinds of fire service areas (*table 1*). Personnel with firefighting experience and knowledge of the fire service areas should be consulted about the weights. Certain FFI categories or classes are generally used in the initial attack section of most simulation models (*table 2*).

EXAMPLE APPLICATION

The following example illustrates how the modified algorithm for determining the FFI list can be applied in any particular fire service area. The initial program level is \$1,162,000, and only initial attack will be modeled.

Table 1—*Possible weights (percent) for firefighting input types in the initial attack program component*

Fire service areas[1]	Firefighting input types[2]																
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
	Percent																
Wilderness																	
FMM_1		30			50			20									
FMM_4								10						10	30	30	20
Grasslands																	
FMM_8				10						20		30		20	20		
Noncommercial forests																	
FMM_{10}				20			10		30			20				20	
Commercial forests																	
FMM_{16}	10									20		20		30			20

[1]Only isolated examples are shown. Number of service areas and FMM's (fire management mixes) will vary according to regional characteristics. FMM's will also vary according to their feasibility.

[2]Numbers 1 through 17 correspond to firefighting input types in *table 2*.

Step 1

Allocate program level dollars to functional activities, assuming the following set of weights (γ):

Functional activity	Weight (γ)
Prevention	0.13
Detection	.10
Initial attack	.54
Fuel management	.23

Hold budget levels for prevention, detection, and fuel management functions constant, and increase the program level.

$$PL = (0.13\ PL + 0.10\ PL + 0.54\ PL + 0.23\ PL)$$

$$PL = (151{,}060_{F_1} + 116{,}200_{F_2} + 627{,}480_{F_3} + 267{,}260_{F_4})$$

Translate the program level into an evaluated program:

$$PL' = \$627{,}480_{F_3}$$

Step 2

To allocate functional activity dollars to firefighting inputs, assume three classes of inputs are in the initial attack functional activity: Category I crews, initial attack teams, and medium engines and crews. The same process can be used when developing production functions for the other activities. Using equation 5, allocate the evaluated program budget to the different inputs, using the following weights (α) for the three inputs:

Input	Weight (α)
Category I crews (B_{31})	0.50
Initial attack teams (B_{32})	.25
Medium engines and crews (B_{33})	.25

Use equation 5 to solve for

$$B_{31} = (0.50)\ (\$627{,}480) = \$313{,}740$$

$$B_{32} = (0.25)\ (\$627{,}480) = \$156{,}870$$

$$B_{33} = (0.25)\ (\$627{,}480) = \$156{,}870$$

Step 3

Translate dollar allocation into number of FFI_{ij} units purchased for the season, using equation 10. The following additional information is needed: unit cost of each FFI, length of workday, length of fire season, and the fraction of the pool (Ω_{ij}) paid by the service area. For this example, the workday and fire season length are the same for all inputs in the initial attack, event though in reality, they may differ:

Input	FFI unit cost	Workday	Fire season length	Seasonal cost	Ω_{ij}
	Dollars/hour	*Hours*	*Days*	*Dollars*	
B_{31}	257	8	100	205,600	0.25
B_{32}	46	8	100	36,800	1.00
B_{33}	55	8	100	44,000	1.00

Use equation 10 to solve for

$\Psi_{31} = (\$313{,}740/\$205{,}600)\ 1/.25 = 6.10$ Category I crew units

$\Psi_{32} = (\$156{,}870/\$36{,}800)\ 1 = 4.26$ initial attack teams

$\Psi_{33} = (\$156{,}870/\$44{,}000)\ 1 = 3.57$ medium engine and crew units

Step 4

To correct fractions in input list, apply the rounding rule. The following input list is obtained: 6 Category I crews, 4 initial attack teams, and 4 medium engines and crews.

Table 2—*Seventeen firefighting input types generally used for initial attack in most simulation models, by category and transportation method*

Category	Transportation method		
	Road	Aircraft	
		Rotary wing	Fixed wing
Personnel teams			
Category I crews	1	2	3
Category II to IV crews	4	5	6
Initial attack team	7		
Helitack team		8	
Smoke jumpers			9
Equipment with personnel			
Tanker—small	10		
Tanker—medium	11		
Tanker—small	12		
Tanker—medium	13		
Retardant			
Rotary wing—small		14	
Rotary wing—medium		15	
Fixed wing—medium			16
Fixed wing—large			17

Table 3—*Original and revised program levels and fire management mixes (results of step 6 for the example application)*

Item	Units	Initial	Revised	Change
Program level comparison				*Pct*
Program level	—	$1,162,000	$1,166,120	0.35
Program budget	—	$ 627,480	$ 631,600	.66
Fire management mix comparison				
Functional activity (γ)				
Prevention	—	0.13	0.129	−.30
Detection	—	.10	.099	−.40
Initial attack	—	.54	.541	.31
Fuels management	—	.23	.228	−.48
Firefighting input list (α)				
Category I crew	6	.50	.488	−2.34
Category II crew	4	.25	.233	−6.76
Engine and crew	4	.25	.278	−11.48

Step 5

Revise program level and fire management mix to reflect the increase (decrease) in cost and change in fire management emphasis. The new program level is $1,166,120 and is distributed as follows:

$$PL = (151{,}060_{F_1} + 116{,}200_{F_2} + 631{,}600_{F_3} + 267{,}260_{F_4})$$

F_1 is allocated for prevention, F_2 for detection, and F_4 for fuel management, all of which remain constant in the PL evaluation. To obtain the initial attack total dollars F_3, multiply the number of firefighting units paid for in the adjusted input list (U_{ij}) by the input unit season cost (SC_{ij}) and add overall firefighting inputs by functional activity (eq. 7).

The set of weights (γ) attached to the new PL remains the same. The rounding rule leads to a new set of weights (α) for firefighting input dollar distribution:

$\alpha_{31} = 0.49$, $\alpha_{32} = 0.23$, $\alpha_{33} = 0.28$.

To obtain the new set of γ's and α's (the FMM), divide the functional activity total (F_i) by the new PL total for the γ's ($\gamma_i = PL'/F_i$), and divide the FFI_{ij} total (B_{ij}) by the functional activity total (F_i) for the α's ($\alpha_{ij} = B_{ij}/F_i$).

Step 6

Compare the revised and original program levels and fire management mixes. This comparison shows that the change is less than the critical threshold of 10 percent (*table 3*); therefore, assume the same FMM. After the whole process is completed, the adjusted program level and new fire management mix will be evaluated in the simulation model to trace one point on a C+NVC curve.

Step 7

Increase the fire management program budget to the next program level and repeat the procedure.

For each fire service area evaluated, the following information is needed:

- Initial program level (PL in dollars)
- Availability cost for nonfire funded personnel
- Initial FMM for the functional activities
  - γ_i's (percent)
  - F_i (dollars)
- Initial FMM for each FFI_{ij}
  - α_{ij}'s (percent)
  - C_{ij} (dollars/hour)
  - W_{ij} (hours/day)
  - S_{ij} (days)
  - Ω_{ij} (percent)

CONCLUSIONS

A procedure[1] that automatically translates program budget levels into fire management mixes is described. The algorithm is general enough to be used with any economic efficiency simulation model now in operation for evaluating fire program options. Any organization that needs to determine a list of resources to perform different tasks within separate functional activities can use this procedure; however, it is most efficient when used for planning at the regional, state, or forest level. The number of program levels that can be tested at once is unlimited.

[1]The computer software for this procedure is written in Fortran 77. Information about this program is available from Armando Golzález Cabán, Pacific Southwest Forest and Range Experiment Station, 4955 Canyon Crest Drive, Riverside, CA 92507.

APPENDIX: FIREFIGHTING INPUT LISTS

To test the Forest Service's Fire Economics Evaluation System (FEES), we developed three lists of firefighting inputs. The different lists show the range of fire management mixes with emphasis on (A) historical program levels, (B) use of air attack rather than ground crews, and (C) use of ground crews rather than air attack. The distributions of the firefighting inputs in the lists represent the percentage of the initial attack program dollars used for the different firefighting inputs after subtracting a fixed amount to cover the availability cost of 3 nonfire funded Category II crews and 17 project crews. Other presuppression expenditures were held constant as follows: prevention—$86,190, detection—$53,040, fuel management—$125,970.

A—Historical Emphasis

Firefighting inputs	Historical[1]		Historical minus 50 pct[2]		Historical plus 50 pct[3]		Historical plus 100 pct[4]	
	No.	Distribution	No.	Distribution	No.	Distribution	No.	Distribution
		Percent		*Percent*		*Percent*		*Percent*
Category II crew	3		3		3		3	
Project crew	17		17		17		17	
Category I crew	1	0.10	1	0.22	2	0.12	2	0.09
Helitack (5 teams and helicopter)	1	.29	0	.00	2	.33	2	.26
Engines—small	3	.15	2	.22	5	.15	7	.16
Engines—medium	2	.15	1	.16	3	.13	5	.17
Bulldozer—medium	1	.05	1	.10	2	.05	2	.04
Airtanker—medium	3	.05	2	.06	5	.04	8	.05
Jumpers (4 teams and aircraft)	2	.21	1	.24	3	.18	5	.23
Total	33	—	28	—	42	—	51	—

[1]Program level—$661,619, initial attack—$396,419.
[2]Program level—$489,101, initial attack—$223.901.
[3]Program level—$903,550, initial attack—$638,351.
[4]Program level—$1,066,935, initial attack—$801,736.

B—Air Forces Emphasis

Firefighting inputs	Historical[1]		Historical minus 50 pct[2]		Historical plus 50 pct[3]		Historical plus 100 pct[4]	
	No.	Distribution	No.	Distribution	No.	Distribution	No.	Distribution
		Percent		*Percent*		*Percent*		*Percent*
Category II crew	3		3		3		3	
Project crew	17		17		17		17	
Category I crew	0	0.00	0	0.00	1	0.06	1	0.04
Helitack (5 teams and helicopter)	2	.57	1	.58	3	.51	4	.50
Engines—small	0	.00	0	.00	0	.00	0	.00
Engines—medium	1	.08	1	.15	2	.09	3	.10
Bulldozer—medium	0	.00	0	.00	0	.00	0	.00
Airtanker—medium	3	.04	2	.06	5	.04	8	.05
Jumpers (4 teams and aircraft)	3	.31	1	.21	5	.30	7	.31
Total	29	—	25	—	36	—	43	—

[1]Program level—$666,083, initial attack—$400,883.
[2]Program level—$501,197, initial attack—$225,997.
[3]Program level—$892,822, initial attack—$627,623.
[4]Program level—$1,091,847, initial attack—$826,648.

C—Ground Forces Emphasis

Firefighting inputs	Historical[1]		Historical minus 50 pct[2]		Historical plus 50 pct[3]		Historical plus 100 pct[4]	
	No.	Distribution	No.	Distribution	No.	Distribution	No.	Distribution
		Percent		*Percent*		*Percent*		*Percent*
Category II crew	3		3		3		3	
Project crew	17		17		17		17	
Category I crew	1	0.10	1	0.20	2	0.13	3	0.13
Helitack (5 teams and helicopter)	0	.00	0	.00	0	.00	0	.00
Engines—small	7	.36	4	.41	11	.35	16	.37
Engines—medium	5	.38	2	.30	7	.33	11	.37
Bulldozer—medium	1	.05	1	.09	2	.06	2	.04
Airtanker—medium	0	.00	0	.00	0	.00	0	.00
Jumpers (4 teams and aircraft)	1	.11	0	.00	2	.13	2	.09
Total	35	—	28	—	44	—	54	—

[1]Program level—$660,162, initial attack—$394,962.
[2]Program level—$503,634, initial attack—$228,434.
[3]Program level—$856,794, initial attack—$591,594.
[4]Program level—$1,070,057, initial attack—$804,858.

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Evaluating economic efficiency of fire management program options requires information on the firefighting inputs, such as vehicles and crews, that would be needed to execute the program option selected. An algorithm was developed to translate automatically dollars allocated to type of firefighting inputs to numbers of units, using a set of weights for a specific fire service area. Solutions that call for fractions of inputs are resolved by a set of rules on rounding out values. The procedure is general enough to be used with any economic efficiency simulation model for evaluating fire program options; however, it is most efficient when used for planning at the regional, state, or forest level.

Retrieval Terms: fire management mix, firefighting inputs, cost plus net value change

GPO 686-442/58233

United States
Department of
Agriculture

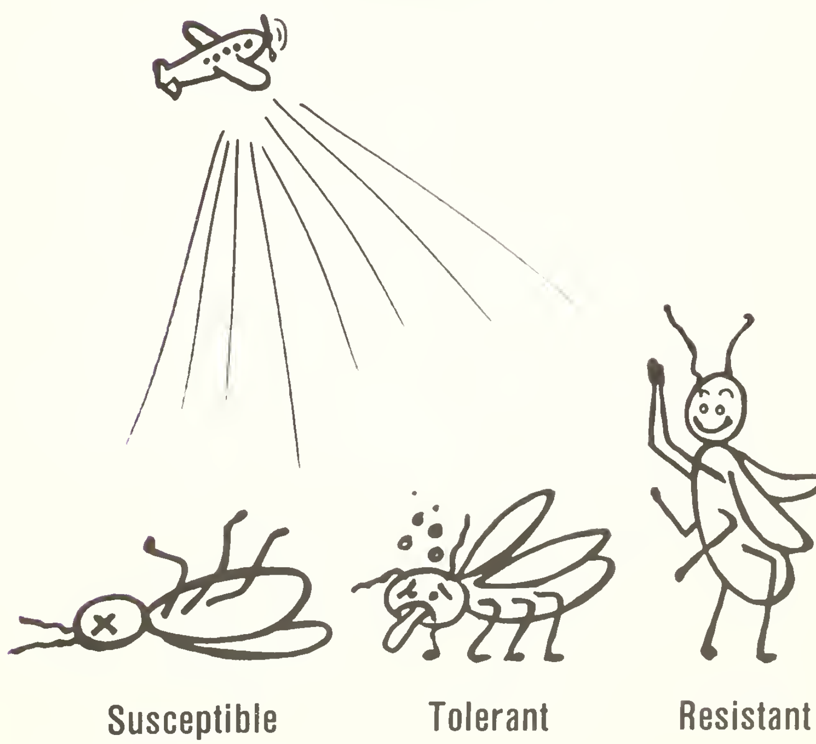
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-89

Computer Prediction of Insecticide Efficacy for Western Spruce Budworm and Douglas-fir Tussock Moth

Jacqueline L. Robertson Molly W. Stock



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Acknowledgments:

We thank Jo Ellen Force, University of Idaho, Moscow; Robert W. Campbell, Forest Service, Corvallis, Oregon; Carroll B. Williams, Forest Service, Berkeley, California; and Michael Saunders, Texas A&M University, College Station, for their helpful reviews of this manuscript. Michael Shook, Washington State University, Pullman, provided valuable assistance with the graphics of the computer program developed in the study reported herein.

Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

April 1986

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CONTENTS

INTRODUCTION

Because laboratory experiments are relatively inexpensive compared to field trials, many attempts have been made to use laboratory data to estimate the results of applying insecticides to populations of a target forest defoliator. Predictions based on well-designed laboratory tests and validated by field tests can help ensure that the desired insect mortality is attained with minimum cost.

Early laboratory studies emphasized screening of candidate insecticides by topical application (e.g., Robertson and Lyon 1973; Robertson and others 1975, 1976). By comparing the relative contact toxicities of new candidates to the toxicities of insecticides already field-tested, crude extrapolations were made to suggest field application rates for each new insecticide. This procedure is inherently inaccurate: measurement units in topical application bear little relationship to those units actually reaching the insect in a field application. Laboratory spray chambers (e.g., Potter 1952, Nigam 1975, Robertson and others 1979) were also used to determine contact effectiveness. Since sprays in these chambers were usually applied to fully exposed insects, even the fact that application rates were calculated in units equivalent to those applied in the field did not improve the accuracy of extrapolations. Field application rates were consistently underestimated because factors such as evaporation and shielding of the insects by foliage were not taken into account. In order to compensate for underestimation of the dose actually reaching the insect in the field, a laboratory-based spray dose estimate was routinely multiplied by an arbitrarily chosen factor of 3 to obtain a field application rate (Williams 1973).

Besides contact effectiveness, toxicity by ingestion was also recognized as an important component of overall efficacy. Bioassays were developed to test toxicity by feeding (e.g., Granett and Retnakaran 1977, Gillette and others 1978) but data were useful only for determination of relative toxicities and were subject to the same limitations as topical and spray application data.

The first deliberate attempt to integrate data on contact and feeding effectiveness in order to predict field effectiveness was a series of bioassays and ranking criteria developed for western spruce budworm (*Choristoneura occidentalis* Freeman) (Robertson and Haverty 1981). Although an improvement over previous attempts such as that of Robertson and Boelter (1979a,b), this method still provided no satisfactorily accurate means to predict field application rates. Subsequently, multiplication factors for suggesting field application rates for western spruce budworm were derived (Haverty and Robertson 1982). These factors were based on a new type of bioassay in which larvae were sprayed while foraging on host plant foliage.

Definition of the relationship between laboratory and field data provided a rational basis for a computer model, based on laboratory data, to predict the efficacy of insecticides applied to the Douglas-fir tussock moth (*Orgyia pseudotsugata* [McDunnough]) and the western spruce budworm (Force and others 1982). The model was used to simulate the effects of applying a given rate of either acephate (Orthene 75S formulation) or carbaryl (Sevin-4-oil formulation) at any time during larval development of either species in the field. The original computer program was not interactive; it calculated single time-point predictions based on fixed instar distributions and separate estimates of contact and feeding mortalities.

This report describes a generalized, interactive version of the computer model that has been expanded to simulate efficacy, over seasonal development of western spruce budworm and Douglas-fir tussock moth, of any insecticide for which the user has laboratory-based concentration-response data. In addition, the model can be used to predict the effect of an insecticide for a particular instar distribution specified by the user. The program has four options, is written in BASIC, and can be operated on a microcomputer. The program listing is found in the *appendix*. Users can obtain a printout of program output by linking the program output to a graphics terminal.

1. MATHEMATICAL PROCEDURES

As in the original model (Force and others 1982), logic of the general model is based on probability theory. The model uses the multiplication law for independent events and the definition of conditional probability for two events. According to the multiplication law, if events A and B are statistically independent, the probability of their joint occurrence equals the product of their individual probabilities. For example, the probability of mortality (M) of an insect in instar i (I_i) when the population is sprayed on day y is the conditional probability $P(M|I_i)$. Once the user has specified the insect species, the genetic response category, the insecticide, and the amount (dose or concentration) reaching the insects, the probability of mortality is equal to the probability of mortality in instar i times the probability of the insect being in instar i, summed

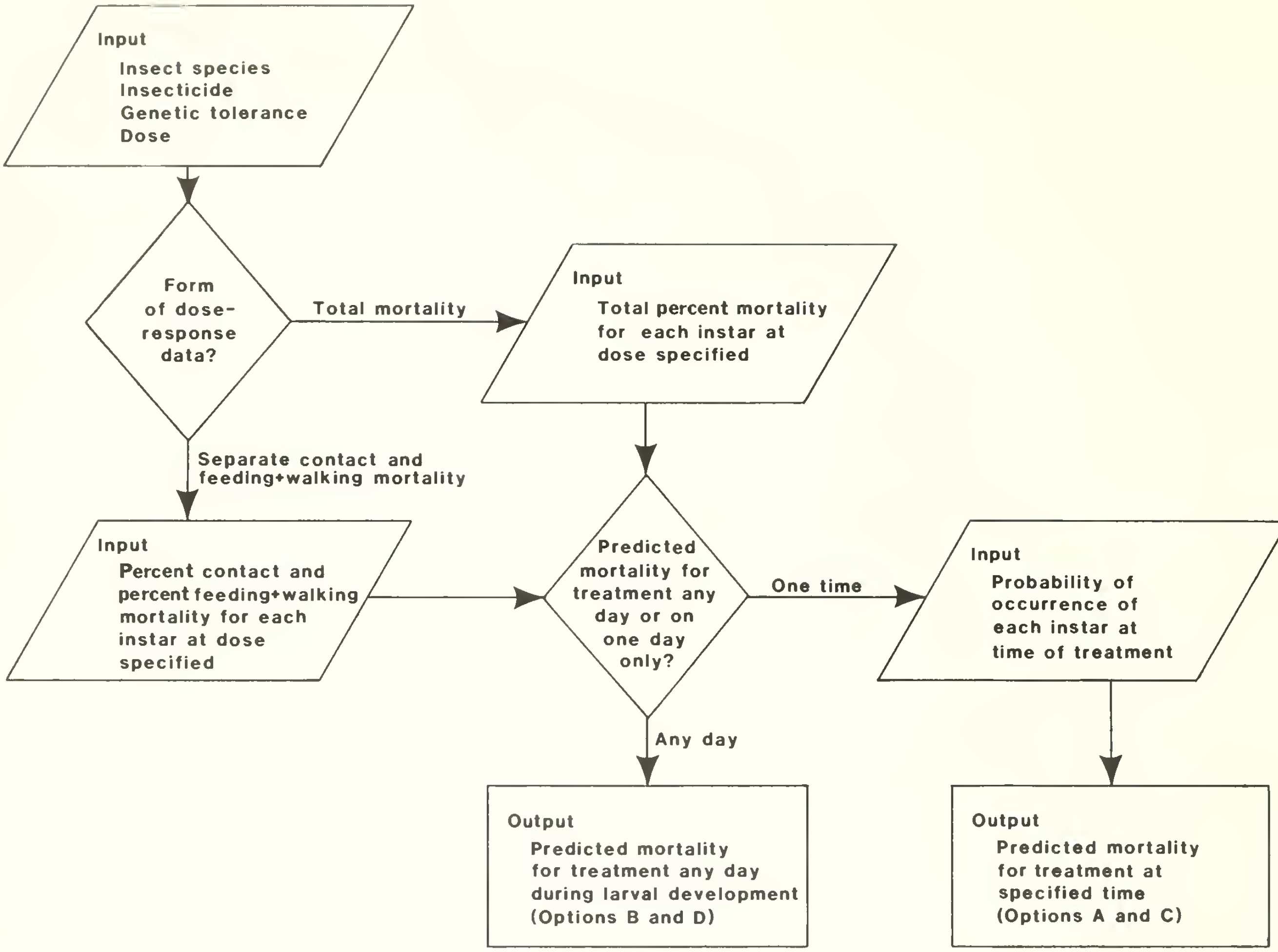

Figure 1—Model logic showing input required and resulting output.

over all instars. Program options (*fig. 1*) are designed to provide (1) a simulation of mortality when the chemical is applied at the specified rate on any day during population development, or (2) a prediction of mortality at a specified time during population development. Two general types of laboratory data can serve as input: (1) separate estimates of mortality by feeding and by contact or (2) estimated total mortality resulting from both feeding and contact. The resultant four options provide the flexibility needed for efficient use of the results of contemporary bioassays.

2. PROGRAM

2.1 Insect

The user first specifies the target species, either Douglas-fir tussock moth (DFTM) or western spruce budworm (WSBW).

2.2 Genetic Predisposition of Population

Responses of populations of both western spruce budworm and Douglas-fir tussock moth to a given chemical may vary widely (Robertson and others 1978; Stock and Robertson 1979, 1980). These differences should be considered in developing realistic models of chemical efficacy. A genetic basis has been documented for differential population responses to acephate and carbaryl (Stock and Robertson 1979, 1980).

A particular population can be designated as relatively tolerant, susceptible, or resistant by comparing its response to that of a standard reference population. One way to separate groups is by comparison of LC_{50}'s and their 95 percent confidence limits (e.g., Roush and Wolfenbarger 1985). A more precise way to separate groups is to use the likelihood ratio test for equality of response (Savin and others 1977). Although the relationships between genetic characteristics and response to other chemicals are at present unknown, the terminology for responses to acephate and carbaryl has been retained in the general model.

2.3 Insecticide

The user must specify the name of the chemical for which a prediction is desired.

2.4 Application Rate Per Unit Area

The user must specify the application rate of each chemical in terms of weight of the active ingredient per unit area. The units specified must match the units of the data base. For example, the application rate is called "dose in unit area" in the program.

In simulation operations (options A and C in the program), the user may compare the predicted effects of applying a given rate on all days over the entire course of population development. Because relative efficacy is of interest in this type of situation, the choice of rate may be entirely arbitrary. To predict efficacy at one point in time (options B and D), the rate may be considered either desirable or likely to reach the insects.

3. PROGRAM OPTIONS

3.1 Option A

This option simulates mortality (over time) based on percent mortality of each instar for a given chemical and rate (user specifies mortality by contact and by feeding) (*figs. 2,3*).

The input data required are (1) percent mortalities for each instar sprayed with a given rate when larvae are fully exposed, and (2) percent mortalities for each instar feeding and walking on foliage sprayed at this rate. These data may be read directly from dose-mortality regression lines for full exposure and feeding plus walking. In the example (*fig. 2*), calculations are made with the Douglas-fir tussock moth population instar distribution stored in the program in order to estimate percent mortality by contact, by feeding plus walking, and total mortality.

We assume that an insect may die from either direct contact with the spray or from feeding and walking on sprayed foliage. The probability of death by direct contact is assumed to depend on the insect's body size (e.g., Busvine 1971); body size is assumed to be a function of the instar at the time of spray. Therefore, mortality by direct contact (MDC) of an insect in instar i is multiplied by the probability of contact (C) for the instar on day y when the toxicant is applied. This probability is summed over all instars (n) to determine expected mortality from direct contact if the treatment were applied on day y:

$$P(MDC_y) = \sum_{i=1}^{n} P(MDC|I_i, C)\ P(C|I_i) \cdot P(I_i)$$

Results of these calculations are shown in the second column of output (CONTACT) (*fig. 2*).

Death from feeding plus walking may occur in insects that survive direct contact effects or that were not hit directly by spray droplets. Therefore, expected mortality from direct contact (MDC) must be subtracted from the total proportion of the population in instar i on day y; the remaining insects in instar i are assumed to be exposed to the chemical by feeding plus walking. The probability of feeding plus walking exposure is then multiplied by the probability of death from feeding plus walking when the insect is in instar i[P(MFW I_i)]. Total probability of mortality from feeding plus walking beginning on day y is the sum of MFW over all instars (n):

$$P(MFW_y) = \sum_{i=1}^{n} P(MFW|I_i) \cdot P(I_i) \cdot [1-P(C|I_i)P(MDC|I_i, C)]$$

Results of these calculations are shown in the third column of output (FEEDING) in (*fig. 2*). Total mortality of the population, if the treatment is applied on day y, is the sum of mortality from both types of exposure:

$$P(TM_y) = \sum_{i=1}^{n} P(MDC|I_i, C)\ P(C|I_i)\ P(I_i) + \sum_{i=1}^{n} P(MFW|I_i) \cdot P(I_i) \cdot [1-P(C|I_i)P(MDC|I_i, C)]$$

Results of these calculations are shown in the fourth column of output (TOTAL) in *figure 2*. The results are displayed in *figure 3*.

3.2 Option B

This option predicts mortality (at one time) based on percent mortalities of each instar for a given chemical and application rate (user specifies mortality by contact and by feeding plus an estimate of the relative proportions of the instars at the time of treatment).

Calculations are performed as described for option A after the user has entered percent mortality for each instar by contact at the application rate, percent mortality by feeding plus walking at the application rate, and the probability of occurrence of each instar.

In the example (*fig. 4*), the user wanted to estimate total mortality when a susceptible western spruce budworm population is sprayed with 53 g/ha of acephate and the population consists of 5 percent second instars, 14 percent third instars, 13 percent fourth instars, 43 percent fifth instars, and 25 percent sixth instars. Data on instar distribution are usually collected by sampling the target population immediately before spray application.

3.3 Option C

This option simulates mortality (over time) based on total percent mortality of each instar for the given chemical and application rate (overall effects of contact and feeding determined from laboratory bioassays).

```
SPECIFY INSECT -- DFTM OR WSBW? DFTM

SPECIFY GENETIC PREDISPOSITION OF POPULATION --
   SUSCEPTIBLE, TOLERANT, OR RESISTANT? TOLERANT

CHEMICAL NAME? ACEPHATE

DOSE IN UNIT/AREA (FOR EXAMPLE, 2.0 G/HA)? 17.7 G/HA

DO YOU WANT A DESCRIPTION OF PROGRAM OPTIONS? YES

PROGRAM OPTIONS ARE:

  A. SIMULATION OF MORTALITY (OVER TIME) BASED ON
     PERCENT MORTALITY OF EACH INSTAR FOR THE GIVEN CHEMICAL
     AND DOSE BY CONTACT AND BY FEEDING DETERMINED FROM
     LABORATORY BIOASSAYS.

  B. PREDICTION OF MORTALITY (FOR ONE TIME) BASED ON
     PERCENT MORTALITY OF EACH INSTAR FOR THE GIVEN CHEMICAL
     AND DOSE BY CONTACT AND BY FEEDING DETERMINED FROM
     LABORATORY BIOASSAYS AND AN ESTIMATE OF THE RELATIVE
     PROPORTIONS OF THE INSTARS AT THE TIME OF TREATMENT.

  C. SIMULATION OF MORTALITY (OVER TIME) BASED ON TOTAL
     PERCENT MORTALITY OF EACH INSTAR FROM THE GIVEN
     CHEMICAL AND DOSE DETERMINED FROM LABORATORY BIOASSAYS.

  D. PREDICTION OF MORTALITY (FOR ONE TIME) BASED ON
     PERCENT MORTALITY OF EACH INSTAR FROM THE GIVEN
     CHEMICAL AND DOSE DETERMINED FROM LABORATORY BIOASSAYS
     AND AN ESTIMATE OF THE RELATIVE PROPORTIONS OF
     INSTARS AT THE TIME OF TREATMENT.

SPECIFY OPTION -- A, B, C, OR D? A

INPUT PERCENT MORTALITY BY CONTACT
    FOR EACH INSTAR AT THIS DOSE.
CONTACT MORTALITY FOR INSTAR 1 ? 60
CONTACT MORTALITY FOR INSTAR 2 ? .5
CONTACT MORTALITY FOR INSTAR 3 ? 6.5
CONTACT MORTALITY FOR INSTAR 4 ? .2
CONTACT MORTALITY FOR INSTAR 5 ? 1.4
CONTACT MORTALITY FOR INSTAR 6 ? 2
VALUES ENTERED WERE: 60  .5  6.5  .2  1.4  2 -- IS THIS CORRECT? YES

INPUT PERCENT MORTALITY BY FEEDING + WALKING
   FOR EACH INSTAR AT THIS DOSE.
FEEDING + WALKING MORTALITY FOR INSTAR 1 ? 20
FEEDING + WALKING MORTALITY FOR INSTAR 2 ? 1
FEEDING + WALKING MORTALITY FOR INSTAR 3 ? .01
FEEDING + WALKING MORTALITY FOR INSTAR 4 ? 13
FEEDING + WALKING MORTALITY FOR INSTAR 5 ? 1.6
FEEDING + WALKING MORTALITY FOR INSTAR 6 ? 11
VALUES ENTERED WERE: 20  1  .01  13  1.6  11 -- IS THIS CORRECT? YES
```

DRY CONDITIONS -- MORTALITY

DFTM ACEPHATE TOLERANT

DOSE 17.7 G/HA

DAY	CONTACT	FEEDING	TOTAL
1	0.08	19.98	20.07
2	0.08	19.98	20.07
3	0.08	19.98	20.07
4	0.08	19.98	20.07
5	0.08	19.98	20.07
6	0.08	19.98	20.07
7	0.07	17.52	17.59
8	0.07	15.62	15.68
9	0.06	13.91	13.97
10	0.05	12.57	12.62
11	0.06	11.58	11.65
12	0.07	10.59	10.67
13	0.08	9.79	9.88
14	0.09	9.18	9.28
15	0.10	8.58	8.68
16	0.11	8.09	8.20
17	0.11	8.06	8.17
18	0.11	8.29	8.40
19	0.11	8.39	8.50
20	0.10	8.70	8.80
21	0.10	8.81	8.92
22	0.10	8.53	8.63
23	0.10	7.96	8.06
24	0.11	7.38	7.49
25	0.11	6.69	6.80
26	0.13	5.95	6.08
27	0.14	5.03	5.17
28	0.15	4.99	5.14
29	0.17	5.01	5.18
30	0.20	5.14	5.34
31	0.22	5.25	5.48
32	0.26	5.49	5.75
33	0.28	5.64	5.92
34	0.31	5.99	6.30
35	0.35	6.31	6.66
36	0.40	6.75	7.15
37	0.45	7.05	7.50
38	0.51	7.33	7.84
39	0.53	7.10	7.63
40	0.56	6.83	7.39
41	0.59	6.47	7.05
42	0.62	6.19	6.81
43	0.66	5.81	6.47
44	0.69	5.65	6.34
45	0.72	5.38	6.10
46	0.76	5.43	6.19
47	0.81	5.69	6.49
48	0.84	5.96	6.80
49	0.89	6.31	7.20
50	0.94	6.78	7.72
51	1.00	7.24	8.23
52	1.05	7.70	8.75
53	1.11	8.26	9.37
54	1.20	9.09	10.29
55	1.27	9.64	10.91
56	1.30	9.92	11.22
57	1.34	10.29	11.63
58	1.40	10.85	12.25
59	1.40	10.85	12.25
60	1.40	10.85	12.25

DO YOU WANT TO TRY ANOTHER OPTION?

Figure 2—Program option A simulates mortality (over time) based on information on percent mortality of each instar for a given chemical and rate. User specifies mortality by contact and by feeding.

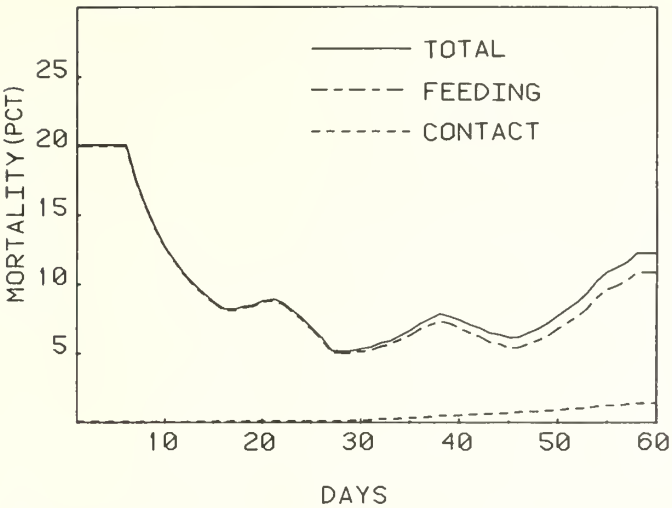

Figure 3—Results of using program option A shows simulation of mortality (contact, feeding, and total) over population development of the Douglas-fir tussock moth.

```
SPECIFY INSECT -- DFTM OR WSBW? WSBW

SPECIFY GENETIC PREDISPOSITION OF POPULATION --
   SUSCEPTIBLE, TOLERANT, OR RESISTANT? SUSCEPTIBLE

CHEMICAL NAME? ACEPHATE

DOSE IN UNIT/AREA (FOR EXAMPLE, 2.0 G/HA)? 53 G/HA

DO YOU WANT A DESCRIPTION OF PROGRAM OPTIONS? NO

SPECIFY OPTION -- A, B, C, OR D? B

INPUT PERCENT MORTALITY BY CONTACT
    FOR EACH INSTAR AT THIS DOSE.
CONTACT MORTALITY FOR INSTAR 1 ? 87
CONTACT MORTALITY FOR INSTAR 2 ? 80
CONTACT MORTALITY FOR INSTAR 3 ? 76
CONTACT MORTALITY FOR INSTAR 4 ? 78
CONTACT MORTALITY FOR INSTAR 5 ? 58
VALUES ENTERED WERE: 87  80  76  78  58 -- IS THIS CORRECT? YES

INPUT PERCENT MORTALITY BY FEEDING + WALKING
   FOR EACH INSTAR AT THIS DOSE.
FEEDING + WALKING MORTALITY FOR INSTAR 1 ? 98
FEEDING + WALKING MORTALITY FOR INSTAR 2 ? 98
FEEDING + WALKING MORTALITY FOR INSTAR 3 ? 98
FEEDING + WALKING MORTALITY FOR INSTAR 4 ? 98
FEEDING + WALKING MORTALITY FOR INSTAR 5 ? 98
VALUES ENTERED WERE: 98  98  98  98  98 -- IS THIS CORRECT? YES


INPUT PROBABILITY (AS DECIMAL -- E.G., .25) OF OCCURRENCE OF
      EACH INSTAR AT TIME OF TREATMENT
PROBABILITY OF INSTAR 1 ? .05
PROBABILITY OF INSTAR 2 ? .14
PROBABILITY OF INSTAR 3 ? .13
PROBABILITY OF INSTAR 4 ? .43
PROBABILITY OF INSTAR 5 ? .25


VALUES ENTERED WERE: .05  .14  .13  .43  .25 -- IS THIS CORRECT? YES
--------------------------------------------
DRY CONDITIONS -- MORTALITY

WSBW          ACEPHATE      SUSCEPTIBLE

          DOSE 53 G/HA

    CONTACT       FEEDING        TOTAL
     16.32         82.01         98.33
--------------------------------------------
DO YOU WANT TO TRY ANOTHER OPTION?
```

Figure 4—Program option B predicts mortality (at one time) based on percent mortality of each instar for a given chemical application rate. User specifies mortality by contact and by feeding plus an estimate of the relative proportions of the instars at the time of treatment.

```
SPECIFY INSECT -- DFTM OR WSBW? WSBW

SPECIFY GENETIC PREDISPOSITION OF POPULATION --
   SUSCEPTIBLE, TOLERANT, OR RESISTANT? TOLERANT

CHEMICAL NAME? UC62644

DOSE IN UNIT/AREA (FOR EXAMPLE, 2.0 G/HA)? 0.71 G/HA

DO YOU WANT A DESCRIPTION OF PROGRAM OPTIONS? NO

SPECIFY OPTION -- A, B, C, OR D? C

INPUT TOTAL PERCENT MORTALITY FOR EACH INSTAR AT THIS DOSE
TOTAL MORTALITY FOR INSTAR 1 ? 65
TOTAL MORTALITY FOR INSTAR 2 ? 65
TOTAL MORTALITY FOR INSTAR 3 ? 57
TOTAL MORTALITY FOR INSTAR 4 ? 89
TOTAL MORTALITY FOR INSTAR 5 ? 90
VALUES ENTERED WERE: 65  65  57  89  90 -- IS THIS CORRECT? YES
--------------------------------------------
DRY CONDITIONS -- MORTALITY

WSBW          UC62644       TOLERANT

          DOSE 0.71 G/HA

DAY   MORTALITY
 1       65.00
 2       65.00
 3       65.00
 4       65.00
 5       65.00
 6       64.84
 7       64.52
 8       64.36
 9       64.60
10       65.40
11       66.28
12       67.16
13       68.12
14       69.32
15       70.60
16       72.22
17       72.37
18       73.43
19       74.16
20       74.98
21       75.56
22       76.79
23       77.79
24       79.28
25       80.53
26       81.53
27       82.78
28       83.94
29       85.35
30       86.52
31       87.10
32       87.69
33       88.28
34       88.96
35       88.99
36       89.35
37       89.39
38       89.76
39       89.80
40       89.85
41       89.89
42       89.93
43       89.97
44       89.99
45       90.00
46       90.00
47       90.00
--------------------------------------------
DO YOU WANT TO TRY ANOTHER OPTION?
```

Figure 5—Program option C simulates mortality (over time) based on total percent mortality of each instar for the given chemical and application rate. User specifies overall effects of contact and feeding as determined from laboratory bioassays.

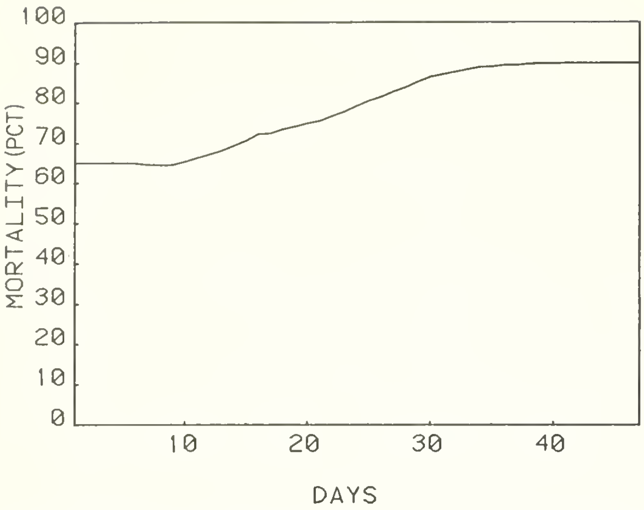

Figure 6—Results using program option C shows total mortality over population development of the western spruce budworm treated with the insecticide UC 62644.

Simulation over the course of population development is done with option C after the user has input total percent mortality read from dose-mortality regressions for insects sprayed while on host plant foliage. This input, P(MCFW)—the probability of mortality by contact, feeding, and walking—is multiplied by the values of $P(I_y)$ contained in the program to simulate $P(TM_y)$ over the course of population development.

In the example (*fig. 5*), the user has specified that a tolerant western spruce budworm population has been sprayed with 0.71 g/ha of the insecticide UC62644. Mortality data for other instars exposed to this application rate are second (65 pct), third (65), fourth (57), fifth (89), and sixth (90). The results of this simulation are graphically illustrated in *figure 6*.

3.4 Option D

This option predicts mortality (at one time) based on percent mortalities of each instar for the given chemical and application rate as determined from laboratory bioassays, and an estimate of the relative proportions of instars present at the time of treatment.

With option D, total mortality is predicted for one time during population development after the user has provided percent mortality for each instar at a given dose and instar distribution (*fig. 7*).

```
SPECIFY INSECT -- DFTM OR WSBW? WSBW

SPECIFY GENETIC PREDISPOSITION OF POPULATION --
   SUSCEPTIBLE, TOLERANT, OR RESISTANT? TOLERANT

CHEMICAL NAME? UC62644

DOSE IN UNIT/AREA (FOR EXAMPLE, 2.0 G/HA)? 0.71 G/HA

DO YOU WANT A DESCRIPTION OF PROGRAM OPTIONS? NO

SPECIFY OPTION -- A, B, C, OR D? D

INPUT TOTAL PERCENT MORTALITY FOR EACH INSTAR AT THIS DOSE
TOTAL MORTALITY FOR INSTAR 1 ? 65
TOTAL MORTALITY FOR INSTAR 2 ? 65
TOTAL MORTALITY FOR INSTAR 3 ? 57
TOTAL MORTALITY FOR INSTAR 4 ? 89
TOTAL MORTALITY FOR INSTAR 5 ? 90
VALUES ENTERED WERE: 65  65  57  89  90 -- IS THIS CORRECT? YES

INPUT PROBABILITY (AS DECIMAL -- E.G., .25) OF OCCURRENCE OF
      EACH INSTAR AT TIME OF TREATMENT
PROBABILITY OF INSTAR 1 ? .05
PROBABILITY OF INSTAR 2 ? .14
PROBABILITY OF INSTAR 3 ? .13
PROBABILITY OF INSTAR 4 ? .43
PROBABILITY OF INSTAR 5 ? .25
VALUES ENTERED WERE: .05  .14  .13  .43  .25 -- IS THIS CORRECT? YES
-------------------------------------------
DRY CONDITIONS -- MORTALITY

WSBW          UC62644        TOLERANT

          DOSE 0.71 G/HA

PERCENT MORTALITY IS 80.53
-------------------------------------------
DO YOU WANT TO TRY ANOTHER OPTION?
```

Figure 7—Program option D predicts mortality (at one time) based on percent mortalities of each instar for the given chemical and application rate as determined from laboratory bioassays, and on estimate of the relative proportions of instars present at the time of treatment.

4. DISCUSSION

Both as a research tool and as an aid for forest managers, this program will be useful to those specifically concerned with western spruce budworm or Douglas-fir tussock moth population management. For example, the predicted efficacy of several insecticides can be compared to select the one that will be the most effective. If a field application does not achieve the desired impact, reasons for the failure can be traced with the aid of model logic.

The logic of this model should be applicable to other forest defoliators, although such adaptations must await the development of comprehensive data bases. Additional laboratory research is required to determine the roles of environmental variables such as temperature, radiation intensity, and host-plant foliage type or quality in responses of western spruce budworm and Douglas-fir tussock moth to chemicals. Once the requisite data are available, the model can be modified to include these factors.

The present database for the model includes results of bioassays of toxicological response of Douglas-fir tussock moth and western spruce budworm that range from very simple to highly elaborate. These bioassays, done over a 15-year period, provide the most comprehensive database available for any forest defoliator and serve as a guide to the sorts of information needed to begin to realistically predict insecticide efficacy on other forest defoliators. The present model predicts the results of actual field applications done under dry conditions with 73 to 95 percent accuracy (Williams and Robertson 1983).

5. APPENDIX: PROGRAM LISTING

```
LIST
10 REM **************** GENERAL MODEL *****************************************
20 DIM D(60),A$(3),B$(4),C$(11),D$(10),E$(15),F$(1),P(60,6),T(6),R(6),Q(6),M(6),S(60),A(60),F(60),V(60),G(60)
30 REM
40 PRINT 'SPECIFY INSECT -- DFTM OR WSBW';
50    INPUT B$
60    PRINT
70 PRINT 'SPECIFY GENETIC PREDISPOSITION OF POPULATION --'
80 PRINT '   SUSCEPTIBLE, TOLERANT, OR RESISTANT';
90     INPUT C$
100    PRINT
110 PRINT 'CHEMICAL NAME';
120    INPUT D$
130    PRINT
140 PRINT 'DOSE IN UNIT/AREA (FOR EXAMPLE, 2.0 G/HA)';
150    INPUT E$
160    PRINT
170 PRINT 'DO YOU WANT A DESCRIPTION OF PROGRAM OPTIONS';
180    INPUT A$
190    IF A$='NO' THEN 430
200    PRINT
210 PRINT 'PROGRAM OPTIONS ARE:'
220    PRINT
230    PRINT '  A. SIMULATION OF MORTALITY (OVER TIME) BASED ON'
240    PRINT '     PERCENT MORTALITY OF EACH INSTAR FOR THE GIVEN CHEMICAL'
250    PRINT '     AND DOSE BY CONTACT AND BY FEEDING DETERMINED FROM'
260    PRINT '     LABORATORY BIOASSAYS.'
270    PRINT
280    PRINT '  B. PREDICTION OF MORTALITY (FOR ONE TIME) BASED ON'
290    PRINT '     PERCENT MORTALITY OF EACH INSTAR FOR THE GIVEN CHEMICAL'
300    PRINT '     AND DOSE BY CONTACT AND BY FEEDING DETERMINED FROM'
310    PRINT '     LABORATORY BIOASSAYS AND AN ESTIMATE OF THE RELATIVE'
320    PRINT '     PROPORTIONS OF THE INSTARS AT THE TIME OF TREATMENT.'
330    PRINT
340    PRINT '  C. SIMULATION OF MORTALITY (OVER TIME) BASED ON TOTAL'
350    PRINT '     PERCENT MORTALITY OF EACH INSTAR FROM THE GIVEN'
360    PRINT '     CHEMICAL AND DOSE DETERMINED FROM LABORATORY BIOASSAYS.'
370    PRINT
380    PRINT '  D. PREDICTION OF MORTALITY (FOR ONE TIME) BASED ON'
390    PRINT '     PERCENT MORTALITY OF EACH INSTAR FROM THE GIVEN'
400    PRINT '     CHEMICAL AND DOSE DETERMINED FROM LABORATORY BIOASSAYS'
410    PRINT '     AND AN ESTIMATE OF THE RELATIVE PROPORTIONS OF'
420    PRINT '     INSTARS AT THE TIME OF TREATMENT.'
430    PRINT
440 PRINT 'SPECIFY OPTION -- A, B, C, OR D';
450    INPUT F$
460    PRINT
470 REM  ***********************************************************************
480 REM READ DATA (BODY SIZE ADJUSTMENT FACTOR AND POPULATION STRUCTURE)
490 FOR Z=1 TO 2
500 IF Z=2 THEN 550
510 REM DFTM (N=INSTARS, Y=DAYS)
520    N=6
530    Y=60
540 GOTO 580
550 REM WSBW (N=INSTARS, Y=DAYS)
560    N=5
570    Y=47
580 REM READ BODY SIZE ADJUSTMENT FACTOR
590    FOR K=1 TO N
600    READ T(K)
610    NEXT K
620 REM READ POPULATION STRUCTURE
630    FOR I=1 TO Y
640    READ D(I)
650    FOR J=1 TO N
660    READ P(I,J)
670    NEXT J
680    NEXT I
```

```
690 IF B$='DFTM' THEN 710
700 NEXT Z
710 REM  *********************************************************************
720 REM OPTIONS -- DATA INPUT
730 IF F$='C' THEN 1410
740 IF F$='D' THEN 1410
750 REM OPTIONS A AND B
760 PRINT 'INPUT PERCENT MORTALITY BY CONTACT'
770 PRINT '    FOR EACH INSTAR AT THIS DOSE.'
780    FOR I=1 TO N
790    PRINT 'CONTACT MORTALITY FOR INSTAR';I;
800    INPUT R(I)
810    NEXT I
820    PRINT 'VALUES ENTERED WERE:';
830    FOR I=1 TO N
840    PRINT R(I);
850    NEXT I
860    PRINT '-- IS THIS CORRECT';
870    INPUT A$
880    IF A$='NO' THEN 780
890    PRINT
900 PRINT 'INPUT PERCENT MORTALITY BY FEEDING + WALKING'
910 PRINT '    FOR EACH INSTAR AT THIS DOSE.'
920    FOR I=1 TO N
930    PRINT 'FEEDING + WALKING MORTALITY FOR INSTAR';I;
940    INPUT Q(I)
950    NEXT I
960    PRINT 'VALUES ENTERED WERE:';
970    FOR I=1 TO N
980    PRINT Q(I);
990    NEXT I
1000   PRINT '-- IS THIS CORRECT';
1010   INPUT A$
1020   IF A$='NO' THEN 920
1030   PRINT
1040   IF F$='A' THEN 1250
1050 REM OPTIONS B AND D
1060 PRINT
1070 PRINT 'INPUT PROBABILITY (AS DECIMAL -- E.G., .25) OF OCCURRENCE OF'
1080 PRINT '       EACH INSTAR AT TIME OF TREATMENT'
1090    I=1
1100    FOR J=1 TO N
1110    PRINT 'PROBABILITY OF INSTAR';J;
1120    INPUT P(I,J)
1130    NEXT J
1140    PRINT 'VALUES ENTERED WERE:';
1150    FOR J=1 TO N
1160    PRINT P(I,J);
1170    NEXT J
1180    PRINT '-- IS THIS CORRECT';
1190    INPUT A$
1200    IF A$='NO' THEN 1090
1210    IF F$='D' THEN 1560
1220 REM OPTION B
1230    J=1
1240    GOTO 1320
1250 REM OPTIONS A AND B
1260 REM CALCULATE ADJUSTED CONTACT MORTALITY (A), ADJUSTED FEEDING
1270 REM       MORTALITY (V), AND TOTAL MORTALITY (G)
1280    FOR J=1 TO Y
1290    A(J)=0
1300    V(J)=0
1310    G(J)=0
1320    FOR K=1 TO N
1330    A(J)=A(J)+(R(K)*T(K)*P(J,K))
1340    V(J)=V(J)+(Q(K)*(P(J,K)-(R(K)*T(K)*P(J,K)/100)))
1350    NEXT K
1360    G(J)=A(J)+V(J)
1370    IF F$='B' THEN 1630
1380    NEXT J
1390    GOTO 1630
1400 REM OPTIONS C AND D
1410 PRINT 'INPUT TOTAL PERCENT MORTALITY FOR EACH INSTAR AT THIS DOSE'
1420    FOR I=1 TO N
1430    PRINT 'TOTAL MORTALITY FOR INSTAR';I;
```

```
1440    INPUT M(I)
1450    NEXT I
1460    PRINT 'VALUES ENTERED WERE:';
1470    FOR I=1 TO N
1480    PRINT M(I);
1490    NEXT I
1500    PRINT '-- IS THIS CORRECT';
1510    INPUT A$
1520    IF A$='NO' THEN 1420
1530 IF F$='D' THEN 1050
1540 REM OPTION C
1550    FOR I=1 TO Y
1560 REM OPTIONS C AND D
1570    G(I)=0
1580    FOR J=1 TO N
1590    G(I)=G(I)+(M(J)*P(I,J))
1600    NEXT J
1610    IF F$='D' THEN 1630
1620    NEXT I
1630 REM  ***********************************************************************
1640 REM PRINTOUT
1650 REM ALL OPTIONS
1660    REM PRINT MORTALILTY FOR SINGLE DOSE LEVEL
1670    PRINT '--------------------------------------------'
1680    PRINT 'DRY CONDITIONS -- MORTALITY'
1690    PRINT
1700    PRINT B$,D$,C$
1710    PRINT
1720    PRINT '            DOSE ';E$
1730    PRINT
1740 IF F$='A' THEN 1780
1750 IF F$='B' THEN 1840
1760 IF F$='C' THEN 1880
1770 IF F$='D' THEN 1940
1780 REM OPTION A
1790    PRINT 'DAY     CONTACT      FEEDING      TOTAL'
1800    FOR J=1 TO Y
1810    PRINT USING ' ##     ###.##       ###.##       ###.##';D(J),A(J),V(J),G(J)
1820    NEXT J
1830    GOTO 1960
1840 REM OPTION B
1850    PRINT '    CONTACT      FEEDING       TOTAL'
1860    PRINT USING '    ###.##       ###.##       ###.##';A(J),V(J),G(J)
1870    GOTO 1960
1880 REM OPTION C
1890    PRINT 'DAY   MORTALITY'
1900    FOR J=1 TO Y
1910    PRINT USING '##        ###.##';D(J),G(J)
1920    NEXT J
1930    GOTO 1960
1940 REM OPTION D
1950    PRINT 'PERCENT MORTALITY IS';G(I)
1960 PRINT '--------------------------------------------'
1970 PRINT 'DO YOU WANT TO TRY ANOTHER OPTION';
1980 INPUT A$
1990 IF A$='YES' THEN RUN
2000 END
2010 REM  ***********************************************************************
2020 REM DFTM BODY SIZE ADJUSTMENT FACTOR
2030 DATA 0.0014,0.011,0.048,0.10,0.26,0.70
2040 REM
2050 REM DFTM POPULATION STRUCTURE, ASYNCHRONOUS
2060 DATA  1,1,0,0,0,0,0
2070 DATA  2,1,0,0,0,0,0
2080 DATA  3,1,0,0,0,0,0
2090 DATA  4,1,0,0,0,0,0
2100 DATA  5,1,0,0,0,0,0
2110 DATA  6,1,0,0,0,0,0
2120 DATA  7,.87,.13,0,0,0,0
2130 DATA  8,.77,.23,0,0,0,0
2140 DATA  9,.68,.32,0,0,0,0
2150 DATA 10,.61,.38,0,0,0,0
2160 DATA 11,.56,.39,.05,0,0,0
2170 DATA 12,.51,.40,.09,0,0,0
2180 DATA 13,.47,.40,.13,0,0,0
2190 DATA 14,.44,.39,.17,0,0,0
```

```
2200 DATA 15,.41,.38,.21,0,0,0
2210 DATA 16,.38,.36,.25,.01,0,0
2220 DATA 17,.36,.34,.26,.04,0,0
2230 DATA 18,.34,.32,.25,.09,0,0
2240 DATA 19,.32,.30,.25,.13,0,0
2250 DATA 20,.31,.29,.23,.17,0,0
2260 DATA 21,.29,.27,.22,.21,.01,0
2270 DATA 22,.25,.27,.22,.25,.01,0
2280 DATA 23,.20,.27,.22,.28,.03,0
2290 DATA 24,.15,.27,.22,.31,.05,0
2300 DATA 25,.10,.27,.22,.33,.08,0
2310 DATA 26,.06,.27,.22,.33,.12,0
2320 DATA 27,.01,.27,.22,.33,.17,0
2330 DATA 28,0,.23,.22,.34,.21,0
2340 DATA 29,0,.19,.22,.34,.25,0
2350 DATA 30,0,.13,.22,.34,.30,.01
2360 DATA 31,0,.09,.22,.34,.33,.02
2370 DATA 32,0,.06,.20,.34,.36,.04
2380 DATA 33,0,.04,.17,.34,.40,.05
2390 DATA 34,0,.03,.13,.34,.42,.08
2400 DATA 35,0,.01,.11,.34,.43,.11
2410 DATA 36,0,0,.07,.34,.44,.15
2420 DATA 37,0,0,.04,.33,.44,.19
2430 DATA 38,0,0,.01,.31,.44,.24
2440 DATA 39,0,0,0,.28,.47,.25
2450 DATA 40,0,0,0,.24,.49,.27
2460 DATA 41,0,0,0,.20,.52,.28
2470 DATA 42,0,0,0,.16,.54,.30
2480 DATA 43,0,0,0,.11,.57,.32
2490 DATA 44,0,0,0,.08,.58,.34
2500 DATA 45,0,0,0,.04,.60,.36
2510 DATA 46,0,0,0,.02,.59,.39
2520 DATA 47,0,0,0,.01,.56,.43
2530 DATA 48,0,0,0,.01,.53,.46
2540 DATA 49,0,0,0,0,.49,.51
2550 DATA 50,0,0,0,0,.44,.56
2560 DATA 51,0,0,0,0,.39,.61
2570 DATA 52,0,0,0,0,.34,.66
2580 DATA 53,0,0,0,0,.28,.72
2590 DATA 54,0,0,0,0,.19,.81
2600 DATA 55,0,0,0,0,.13,.87
2610 DATA 56,0,0,0,0,.10,.90
2620 DATA 57,0,0,0,0,.06,.94
2630 DATA 58,0,0,0,0,0,1
2640 DATA 59,0,0,0,0,0,1
2650 DATA 60,0,0,0,0,0,1
2660 REM
2670 REM WSRW BODY SIZE ADJUSTMENT FACTOR
2680 DATA 0.0033,0.013,0.05,0.16,0.71
2690 REM
2700 REM WSRW POPULATION STRUCTURE, ASYNCHRONOUS
2710 DATA  1,1,0,0,0,0
2720 DATA  2,.99,.01,0,0,0
2730 DATA  3,.97,.03,0,0,0
2740 DATA  4,.88,.12,0,0,0
2750 DATA  5,.79,.21,0,0,0
2760 DATA  6,.72,.26,.02,0,0
2770 DATA  7,.65,.29,.06,0,0
2780 DATA  8,.57,.31,.11,.01,0
2790 DATA  9,.51,.32,.14,.03,0
2800 DATA 10,.46,.31,.16,.07,0
2810 DATA 11,.42,.30,.17,.11,0
2820 DATA 12,.39,.28,.18,.15,0
2830 DATA 13,.36,.27,.18,.19,0
2840 DATA 14,.33,.25,.18,.24,0
2850 DATA 15,.31,.23,.17,.29,0
2860 DATA 16,.29,.22,.17,.32,.01
2870 DATA 17,.28,.20,.16,.35,.01
2880 DATA 18,.26,.19,.15,.37,.03
2890 DATA 19,.24,.18,.15,.39,.04
2900 DATA 20,.23,.17,.14,.40,.06
2910 DATA 21,.22,.17,.13,.40,.08
2920 DATA 22,.17,.17,.13,.42,.11
2930 DATA 23,.13,.17,.13,.42,.15
2940 DATA 24,.08,.16,.13,.43,.20
```

```
2950 DATA 25,.05,.14,.13,.43,.25
2960 DATA 26,.03,.12,.13,.43,.29
2970 DATA 27,.01,.09,.13,.43,.34
2980 DATA 28,.01,.07,.11,.43,.38
2990 DATA 29,0,.05,.09,.43,.43
3000 DATA 30,0,.03,.07,.42,.48
3010 DATA 31,0,.02,.06,.42,.50
3020 DATA 32,0,.01,.05,.41,.53
3030 DATA 33,0,0,.04,.40,.56
3040 DATA 34,0,0,.02,.38,.60
3050 DATA 35,0,0,.02,.35,.63
3060 DATA 36,0,0,.01,.32,.67
3070 DATA 37,0,0,.01,.28,.71
3080 DATA 38,0,0,0,.24,.76
3090 DATA 39,0,0,0,.20,.80
3100 DATA 40,0,0,0,.15,.85
3110 DATA 41,0,0,0,.11,.89
3120 DATA 42,0,0,0,.07,.93
3130 DATA 43,0,0,0,.03,.97
3140 DATA 44,0,0,0,.01,.99
3150 DATA 45,0,0,0,0,1
3160 DATA 46,0,0,0,0,1
3170 DATA 47,0,0,0,0,1
3180 REM ##################################################################
Ok
```

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A generalized interactive computer model that simulates and predicts insecticide efficacy, over seasonal development of western spruce budworm and Douglas-fir tussock moth, is described. This model can be used for any insecticide for which the user has laboratory-based concentration-response data. The program has four options, is written in BASIC, and can be operated on a microcomputer.

Retrieval Terms: insecticides, models, western spruce budworm, Douglas-fir tussock moth

United States
Department of
Agriculture

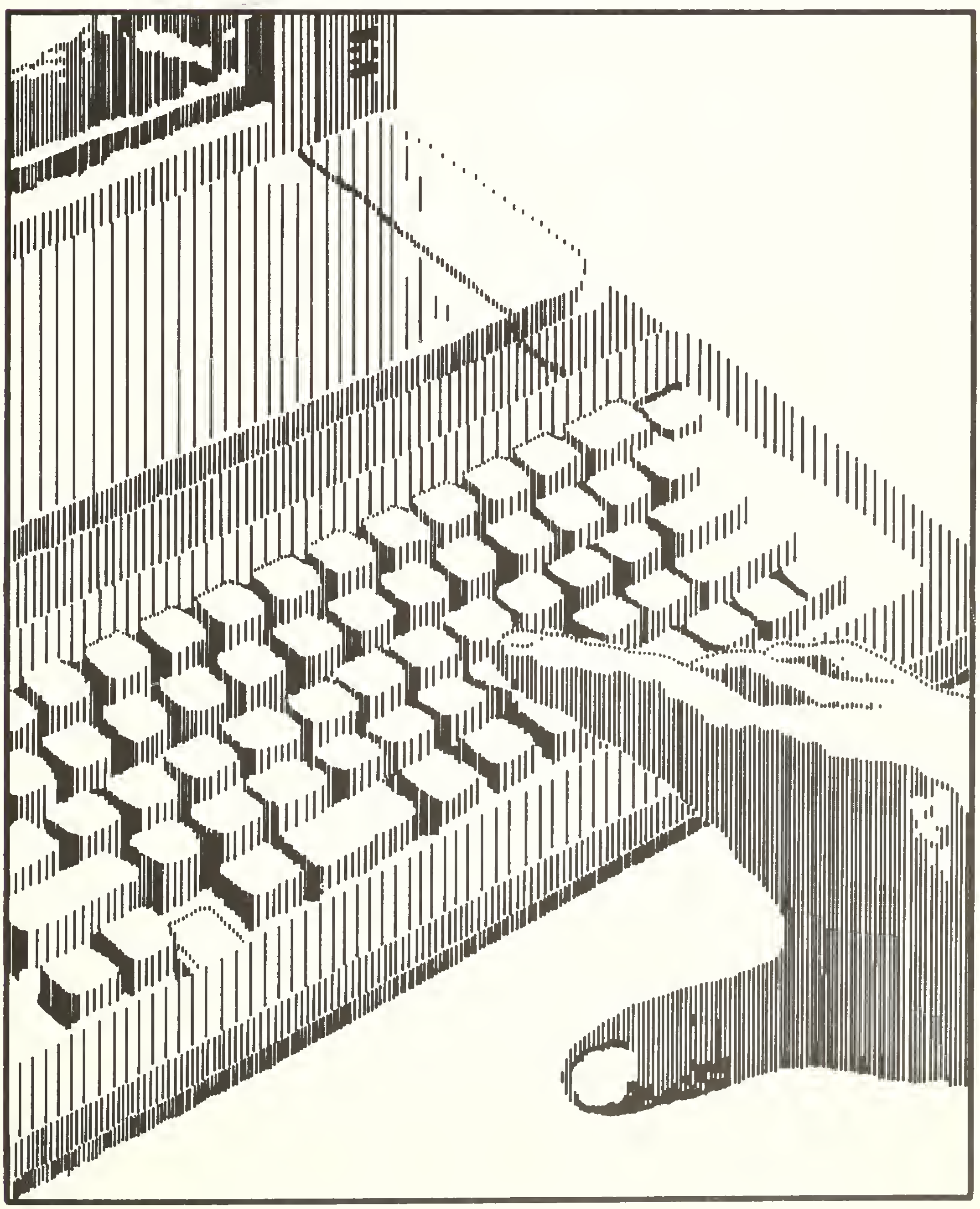
Forest Service

**Pacific Southwest
Forest and Range
Experiment Station**

General Technical
Report PSW-90

Estimating Fire Behavior With FIRECAST: user's manual

Jack D. Cohen



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Publisher:

Pacific Southwest Forest and Range Experiment Station
P.O. Box 245, Berkeley, California 94701

May 1986

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CONTENTS

Preface

I first worked on FIRECAST at the Intermountain Station's Northern Forest Fire Laboratory in Missoula, Montana, in 1978, when there was initial user interest in a national fire behavior program. This original FIRECAST fire behavior program was an expanded version of the southern California FIREMOD computer program.* Despite major improvements to the FIRECAST program and encouraging test operation, official development of the program was stopped in 1979 as a result of funding problems. Dick Harrell of the Forest Service's Pacific Southwest Region (R-5) eventually revived FIRECAST. I began redeveloping the program in 1982, at the Pacific Southwest Station's Forest Fire Laboratory in Riverside, California.

The resulting FIRECAST program offers three fuel model options to users: Northern Forest Fire Laboratory (NFFL) fuel models, National Fire Danger Rating System (NFDRS) fuel models, and southern California brushland (SCAL) fuel models. Each of these three technical routines was checked with other independent sources: the NFFL option was checked against the FIREMOD program at the Northern Forest Fire Laboratory; the NFDRS option was checked against outputs (translated for matching fire behavior) from the NFDRS Administrative and Forest Fire Information Retrieval and Management System (AFFIRMS);[†] and the SCAL option was checked against an independently written program at Colorado State University—a Station cooperator.

Beginning in June 1982, the FIRECAST program and a preliminary user manual were available for operational testing by Federal, State, and local agencies within the State of California. After one season, test users indicated a few operational program changes, which I made for the second season of operational testing in 1983. Users indicated a high level of satisfaction with FIRECAST, and none indicated difficulty in obtaining results.

*Van Gelder, Randall J. **A fire potential assessment model for brush and grass fuels.** Fire Manage. Notes 37(3): 14-16; 1976.

[†]Helfman, Robert S.; Straub, Robert J.; Deeming, John E. **Users' guide to AFFIRMS: time-share computerized processing for fire danger rating.** Gen. Tech. Rep. INT-82. Ogden, UT: Intermountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 1980. 150 p.

1. INTRODUCTION

FIRECAST is a computer program that estimates up to six fire behavior parameters: rate-of-spread, fireline intensity, flame length, perimeter and area, scorch height, and ignition component. Required inputs vary depending on the outputs desired. The program has been operationally tested for use in California.

To obtain a copy of the FIRECAST program software, send your request along with ½-inch computer tape and any specific instructions for writing the tape to Director, Pacific Southwest Forest and Range Experiment Station, P.O. Box 245, Berkeley, CA 94701, Attention: Computer Services Librarian. We can write blocked tapes of line images in either ASCII or EBCDIC of the source codes and data. Or, if you have access to the Forest Service distributed processing system and are connected to the U.S. Department of Agriculture Departmental Network (DEPNET), you can contact SCS:S27A and request instructions for retrieving the FIRECAST files.

This report describes the components of the FIRECAST program and contains the complete operating instructions, which should be read carefully before the program is used. Although FIRECAST only *estimates* fire behavior, the outputs are more reliable when the inputs are estimated carefully.

2. PROGRAM COMPONENTS

Several different models contribute to the computational portion of the FIRECAST program. Following is a summary of the fire model components and their related outputs and required inputs.

2.1 Rate-of-Spread

The Rothermel fire spread model (Rothermel 1972) with modifications by Albini (1976a, b), provides the computational basis for rate-of-spread estimates for all the fuel model options. The rate-of-spread refers to the forward spreading (with the wind and slope), quasi-steady state flaming zone. Required inputs include the physical description of the fuel bed (fuel model), fuel moistures by size class (as related to timelag class), and windspeed and slope in the direction of forward spread.

2.2 Fireline Intensity and Flame Length

The formulations by Byram (1959) form the basis for fireline intensity and flame length. They were adapted by Albini (1976a, b) for use in conjunction with the Rothermel fire spread model. Intermediate calculations from the fire spread model and Anderson's (1969) flaming residence time relation provide the values required to derive Byram's formulation.

2.3 Perimeter and Area

Perimeter and area calculations assume that the fire shape approximates a double ellipse (Anderson 1983) based on empirical fire shape data (Fons 1940). The shape of the double ellipse varies with windspeed in the direction of fire spread. The final perimeter and area estimates require the forward spread distance.

2.4 Scorch Height

Van Wagner (1973) formulated the scorch height equation from a theoretical basis and correlated the theoretical prediction with lethal scorch found in several northeastern tree species. The scorch height equation requires the windspeed and ambient temperature roughly at the flaming level. The fireline intensity requirement is supplied by the calculated fireline intensity.

2.5 Ignition Component

The ignition component, from the National Fire Danger Rating System (Deeming and others 1977), is based on Schroeder's (1969) formulations of the requirements for a theoretical ignition. Assumptions include a sufficiently energetic fire brand and available fine fuels for ignition. The NFDRS treats only the reportable fire situation. Ignitions that do not result in fires sufficient for detection are of no concern. The rate-of-spread serves as a measure of fire reportability: the higher the spread rate, the greater the chance of discovery. Further information on the ignition component can be found elsewhere (Bradshaw and others 1983, Cohen and Deeming 1985). Ambient temperature, fine fuel moisture (1-hr timelag), and the sky cover are required inputs.

3. OPERATING INSTRUCTIONS

In the following instructions, a general format is followed. Program prompts issued by the computer are in **boldface**. In most sections, examples of actual FIRECAST sequences follow the explanation. In these examples, FIRECAST prompts are in boldface and the user response is followed by the new line (carriage return) symbol ([NL]). (Every typed response to a FIRECAST prompt must be followed by pushing the new line (carriage return) key [NL].) The entire FIRECAST example run that appears in *section 4* is an illustration of what will appear on your terminal. This example has no boldface or [NL] symbols.

You access the FIRECAST fire behavior program through your particular system's procedure for accessing and running FORTRAN 77 programs.

3.1 "END"

You may type "END" after any FIRECAST prompt. The program responds with:

CONTINUE FIRECAST (Y/N)?

Typing "N" allows you to exit FIRECAST or to start the prompting sequence again from the beginning. Typing "Y" continues program processing. See *section 3.7 Continuing FIRECAST,* for an explanation of continuing FIRECAST processing.

Example 1—You are designating your display outputs, but desire to interrupt the sequence.

```
DESIGNATE THE OUTPUTS YOU DESIRE FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD
Y [NL]
FIRELINE INTENSITY
END [NL]
CONTINUE FIRECAST (Y/N)?
```

Example 2—You are prompted to input fuel moistures, but you wish to interrupt the sequence.

```
INPUT PERCENT 1HR,10HR,100HR AND LIVE
END [NL]
CONTINUE FIRECAST (Y/N)?
```

3.2 Help Document

Immediately upon entry into FIRECAST, the program prompts:

DO YOU WANT HELP (Y/N)?

Typing "Y" yields a help document that gives a brief explanation of the appropriate responses to FIRECAST prompts. The help document is six, 23-line pages. After each page you have three options:

1. To continue to the next page, type "Y."
2. To exit out of the help document for continued FIRECAST processing, type "N."
3. To exit FIRECAST, type "END."

If you do not want the help document, type "N" and push the new line key [NL]. The next prompt in sequence will be displayed. The help document can be viewed only upon entry to FIRECAST.

[1]Trade names and commercial products are mentioned only for information. No endorsement by the U.S. Department of Agriculture is implied.

3.3 Selecting Outputs

You may choose any or all of the available fire behavior outputs for tabular display. The six available outputs are these: rate-of-spread, fireline intensity, flame length, perimeter and area, scorch height, ignition component.

Rate-of-Spread—The forward rate-of-spread is at the head of the fire front. Rate-of-spread is expressed in chains per hour. To convert to feet per minute, multiply the rate-of-spread by 1.1.

Fireline Intensity—Fireline intensity is the amount of heat in British thermal units released in the flaming front per second, per foot of fire line (Btu/ft/s).

Flame Length—Flame length is defined as the distance from the tip to the base of the burning fuel midway in the flaming front. Flame length is expressed in feet.

Perimeter and Area—Fire perimeter and area represent the fire growth rate assuming an elliptical shape. Perimeter is expressed in chains per hour, and area is expressed in acres per hour squared. To find the growth after a period of time, multiply the displayed perimeter value by the time in hours and multiply the displayed area value by the time in hours, squared. For example, after 2 hours, multiply the displayed acre rate by 4 (2 squared equals 4). Notice that the displayed values for perimeter and area are equal to the growth after 1 hour. *Caution:* do not expect reasonable estimates after long time periods or under highly variable conditions of fuels, weather, and topography.

Scorch Height—Scorch height is the height in feet above the ground where lethal foliage scorch occurs. With temperature as an input, the program calculates scorch height at temperatures other than 77 °F. No temperature factor is needed as with other procedures (Albini 1976a).

Ignition Component—The ignition component is the same as the NFDRS ignition component. It represents the probability that a fire brand will ignite a fire of reportable size.

The program displays:

DESIGNATE THE OUTPUTS YOU DESIRE FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD

If you desire this output, type "Y." If you do not, type "N." After you hit a new line, the next display output on the above list will appear. Continue entering your choices until the list is exhausted. The next prompt will be displayed.

Example 3—You desire all of the fire behavior outputs.

```
DESIGNATE THE OUTPUTS YOU DESIRE FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD
Y [NL]
FIRELINE INTENSITY
Y [NL]
FLAME LENGTH
Y [NL]
PERIMETER AND AREA
Y [NL]
SCORCH HEIGHT
Y [NL]
IGNITION COMPONENT
Y [NL]
```

Example 4—You want only rate-of-spread, flame length, and ignition component.

```
DESIGNATE THE OUTPUTS YOU DESIRE FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD
Y [NL]
FIRELINE INTENSITY
N [NL]
FLAME LENGTH
Y [NL]
PERIMETER AND AREA
N [NL]
SCORCH HEIGHT
N [NL]
IGNITION COMPONENT
Y [NL]
```

3.4 Environmental Site Data

3.4.1 Windspeed

The program prompts:

INPUT 1 = MIDFLAME OR 2 = 20 FT HEIGHT, INITIAL WIND, FINAL WIND, WIND INCREMENT

MIDFLAME corresponds to hand-held windspeed measurements. The 20 FT HEIGHT corresponds to windspeed measured 20 feet above the surface. That surface can be virtually ground level or a closed tree canopy 100 feet above the ground. Input a "1" or "2" for the appropriate windspeed measurement level.

You can select a windspeed range starting with "INITIAL WIND," ending with "FINAL WIND" and incremented by "WIND INCREMENT." The maximum number of wind values in the output table is six. Therefore, if you select a wind range and an increment that results in more than six wind incremental values, your "FINAL WIND" input is recalculated. Input all the windspeeds to the nearest whole number value in miles per hour.

Example 5—You wish to enter midflame windspeeds. The wind range is 0 to 10 mi/h. You wish to see a fire behavior estimate for each 2 mi/h within the range.

```
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
[1,0,10,2] [NL]
```

3.4.2 Percent Slope, Temperature, and Sky Cover

The program prompts:

INPUT PERCENT SLOPE, TEMP, AND SKY COVER

The percent slope used is the conventional slope tangent times 100, to the nearest whole percent. The program assumes that the slope input is in the average wind direction.

The temperature and sky cover inputs apply to the calculation of only the IGNITION COMPONENT and the SCORCH HEIGHT. If you do not desire to output the IGNITION COMPONENT and the SCORCH HEIGHT, you must still input "TEMP" and "SKY COVER" to satisfy format requirements of the computer program.

"TEMP" is the instrument level (4.5 ft above ground) temperature in degrees Fahrenheit, to the nearest whole number.

"SKY COVER" accounts for the increase in fuel temperature above the instrument level temperature due to solar radiation. It applies to cloud or vegetative canopy cover or both.

"SKY COVER" input is the same as the NFDRS code (Cohen and Deeming 1985) describing the sky cover (state of the weather equals 0 – 3):

"SKY COVER"	Sky or canopy coverage
	Percent
0	0 – 10
1	10 – 50
2	50 – 90
3	90 +

Example 6—You wish to enter percent slope, temperature, and sky cover. The percent slope in the direction of the wind is 30 percent. The instrument level (4.5 ft above ground) temperature is 80 °F. The sky is 40 percent cloud covered, and the sky cover code is 1.

```
INPUT PERCENT SLOPE, TEMP, AND SKY COVER
30,80,1 [NL]
```

3.5 Selecting a Fuel Model

FIRECAST offers three fuel model sets: the Northern Forest Fire Laboratory (NFFL) fire behavior fuel models (Rothermel 1983), the National Fire Danger Rating System (NFDRS) fuel models (Cohen and Deeming 1985), and the southern California (SCAL) brush fuel models (Rothermel and Philpot 1973). The NFFL models are those offered by the TI-59 fire behavior module (Burgan 1979) and used for fire behavior predictions.[1] The NFDRS fuel models are included for those managers who are already comfortable selecting these fuel models and find them appropriate for their situations. The NFDRS context and standards do not apply when using these fuel models in FIRECAST. Further explanation of the NFDRS and its component models can be found elsewhere (Cohen 1985). The SCAL fuel models are specifically included for southern California brushland fuels because these models currently provide the best input for predictive modeling of fire behavior in these fuels.

The program prompts:

INPUT FUEL MODEL SOURCE AND FUEL MODEL

You may use any of the three sets of fuel models contained in FIRECAST. The Northern Forest Fire Laboratory (NFFL) set has 13 fuel models, the National Fire Danger Rating (DR78) set has 20 fuel models, and the southern California (SCAL) set has 2 variable brushland fuel models. The required inputs for selecting the fuel model source and fuel model are these:

Fuel model source	Fuel model
NFFL	1-13
DR78	A-L,N-U
SCAL	B,C

Keep in mind that selecting a fuel model is a "best fit" to your actual fuel bed situation. Following are the fuel model designators and the fuel types by source.

NFFL
- 1 Short grass
- 2 Timber/grass understory
- 3 Tall grass
- 4 Mixed brush/heavy dead
- 5 Brush/not chaparral
- 6 Dormant brush
- 7 Southern rough
- 8 Closed timber litter
- 9 Hardwood litter
- 10 Heavy timber litter
- 11 Light logging slash
- 12 Medium logging slash
- 13 Heavy logging slash

DR78
- A Western annual grass
- B California mixed chaparral
- C Pine/grass savanna
- D Southern rough
- E Hardwood litter (winter)
- F Intermediate brush
- G Short-needle conifer (heavy dead)
- H Short-needle conifer (light dead)
- I Heavy slash
- J Medium slash
- K Light slash
- L Western perennial grass
- N Sawgrass
- O High pocosin
- P Southern pine
- Q Alaskan black spruce
- R Hardwood litter (summer)
- S Tundra
- T Sagebrush/grass
- U Western long-needle pine

SCAL
- B Mixed chaparral (nonchamise)
- C Chamise

Example 7—You have decided to use the southern California brushland fuel model (SCAL) for chamise (C).

```
INPUT FUEL MODEL SOURCE AND FUEL MODEL
SCAL,C [NL]
```

3.6 *Fuel Bed Data*

3.6.1 NFFL and DR78 Options

Further prompts for input depend on the fuel model source chosen. If you chose NFFL, then the next (and last) prompt is:

INPUT PERCENT 1HR,10HR,100HR, AND LIVE

1HR represents 1 hour timelag class fuel moisture.
10HR represents 10 hour timelag class fuel moisture.
100HR represents 100 hour timelag class fuel moisture.
LIVE represents average fuel moisture of all living fine fuels less than ¼ inch in thickness or diameter.

Example 8—You chose the NFFL source fuel model. Your fuel moistures by timelag class are these:

1HR = 3 percent
10HR = 5 percent
100HR = 7 percent
LIVE = 100 percent

```
INPUT PERCENT 1HR,10HR,100HR AND LIVE
3,5,7,100 [NL]
```

If instead you chose DR78, then the next prompt will be

INPUT PERCENT 1HR,10HR,100HR,1000HR,HERB AND WOODY

1HR represents 1 hour timelag class fuel moisture.
10HR represents 10 hour timelag class fuel moisture.
100HR represents 100 hour timelag class fuel moisture.
1000HR represents 1000 hour timelag class fuel moisture.
HERB represents the average percent live fuel moisture of all fine herbaceous fuels less than ¼ inch thick or diameter.
WOODY represents the average percent live fuel moisture of all fine woody fuels less than ¼ inch thick or diameter.

As with the NFFL option, after you input the fuel moistures, the program processes the information and displays the results in the output table.

Example 9—You chose the DR78 source fuel model. Your percent fuel moistures by timelag class are these:

1HR = 5 percent
10HR = 7 percent
100HR = 10 percent
1000HR = 13 percent
HERBACEOUS = 120 percent
WOODY = 100 percent

```
INPUT PERCENT 1HR,10HR,100HR,1000HR,HERB AND WOODY
5,7,10,13,120,100 [NL]
```

3.6.2 SCAL Option

If you chose the brush option SCAL, a much different set of prompts appears requesting fuel moistures and the time of the year:

INPUT PERCENT 1-HR FM, 10-HR FM, AND MONTH AND DAY

This prompt is followed by

OPTIONAL USER INPUTS FOR SCAL

then

INPUT PERCENT DEAD AND BRUSH DEPTH (FT):

Input is optional here. FIRECAST limits your inputs to between 10 and 64 percent. Inputs less than 10 percent are set to 10 percent; inputs greater than 64 percent are set to 64 percent. You may enter the inputs as requested, or if you do not have the information, entering "NO" will lead to the prompt,

INPUT BRUSH AGE (YRS):

Caution: recent research indicates that age is not a reliable indicator of chaparral fuel characteristics.

You must enter "NO" to the previous prompt for brush percent dead and depth, if you enter an age. Otherwise, previous inputs for percent dead and depth will be used if available. If not available, an "INPUT ERROR . . ." results. FIRECAST limits your age inputs to a maximum of 55 years. An age input greater than 55 years is set to 55 years.

You must input information to one of these two prompts. The percent dead and depth input option is available to help you provide better brush fuel estimates. Brush age does not describe site-specific brush fuel characteristics nearly as well as estimates of percent dead and brush depth. Therefore, try to use the specific inputs of percent dead and brush depth whenever possible.

The next prompt,

INPUT NEW AND OLD GROWTH LIVE PERCENT FUEL MOISTURES:

requests the live brush moisture contents that correspond to the new and old growth sampled moisture contents. If you do not have site-specific live moisture samples, choose the nearest or most appropriate standard sample site. If you do not have the live moisture information, you must enter "NO." This defaults to the live moisture contents calculated by the program on the basis of the time of the year. (Note: You must enter the month and day in the previous prompt whether or not live moisture contents are entered.) No more prompts appear until display of the outputs is completed.

Example 10—You chose the SCAL source fuel model. Your dead percent fuel moistures are these:

1HR = 4 percent
10HR = 7 percent.
The month and day are SEPTEMBER 15.
You have the following percent dead and brush depth:
PERCENT DEAD = 40
BRUSH DEPTH = 5 feet.
You also have the current new and old growth percent live fuel moistures from the nearest appropriate sampling site:
NEW GROWTH = 60
OLD GROWTH = 50

```
INPUT PERCENT 1HR,10HR, AND MONTH AND DAY
4,7,9,15 [NL]
OPTIONAL USER INPUTS FOR SCAL
INPUT PERCENT DEAD AND BRUSH DEPTH (FT):
40,5 [NL]
INPUT PERCENT NEW AND OLD GROWTH LIVE FUEL MOISTURES:
60,50 [NL]
```

Example 11—You chose the SCAL source fuel model. Your dead fuel percent moisture contents and month and day remain the same as in example 10. However, you do not have information on the percent dead, brush depth, or new and old growth live percent fuel moistures. You do have a brush age estimate of 50 years.

```
INPUT PERCENT 1HR,10HR, AND MONTH AND DAY
4,7,9,15 [NL]
OPTIONAL USER INPUTS FOR SCAL
INPUT PERCENT DEAD AND BRUSH DEPTH (FT):
NO [NL]
INPUT BRUSH AGE (YRS):
50 [NL]
INPUT NEW AND OLD GROWTH LIVE PERCENT FUEL MOISTURES:
NO [NL]
```

3.7 Continuing FIRECAST

Immediately following the output display table or after you type "END," the program prompts:

CONTINUE FIRECAST (Y/N)?

If you desire to end your FIRECAST processing, enter "N." Your response will be answered with

STOP

[your system's prompt]

FIRECAST has returned you to your system's command.

If you desire to continue FIRECAST processing, enter "Y." The prompt,

CHANGE OUTPUT DISPLAY (Y/N)?

follows. You now have the option of changing what fire behavior outputs appear in the display table (from the list of available fire behavior outputs). If you are satisfied with the type of fire behavior options displayed, enter "N."

If you desire an output display change, enter "Y." FIRECAST processing continues with the prompt,

DESIGNATE THE OUTPUTS YOU DESIRE
FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD

You have seen this before; continue as before (*sec. 3.3*). The program processing continues with the prompt,

1 = MIDFLAME OR 2 = 20FT HEIGHT, INITIAL WIND, FINAL WIND,WIND INCREMENT

Once you have initially input the required data and received the output table, you may wish to continue FIRECAST processing. Those lines of data that do not change need not be reentered. Note: This procedure does not apply to changing the output display. You must always enter "Y [NL]" if you want the option displayed. When the prompt requests your data input, pressing the [NL] key will execute the next operation in sequence. The data for the line remains unchanged. This also applies if you have been using several fuel model options. If you return to a previously selected source option, you do not have to reenter data for a line that remains unchanged. This procedure does not apply once you exit the FIRECAST program and then reenter. Also, if an "INPUT ERROR. . ." occurs, the data for that prompt has been disturbed and needs to be reentered. The above procedure replaces the data file procedures of the previous FIRECAST version.

When making subsequent FIRECAST runs, you will probably need to review information previously input. You can display a line of input data by typing "DISPLAY" after the program prompt. Only the previously input information pertaining to the last prompt is displayed. After the input data listing, the last prompt appears below the data list. You can enter new data, leave data unchanged, or exit according to regular procedures. "DISPLAY" does not apply to the selection of the fire behavior output options.

Example 12—You have completed a run and wish to continue FIRECAST processing. You want to display a line of previously input data and then change the data.

```
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
DISPLAY [NL]
                    1                            0            10            2
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
2,0,30,5 [NL]
```

3.8 Unexpected Characters

3.8.1 Appearance of "*" After the Rate-of-Spread Value

You may see an asterisk (*) with the displayed rate-of-spread values. You will also notice that the rate-of-spread values as well as other values (except scorch height) remain constant. This condition indicates that the windspeeds for which the * appeared are at or above maximum for the reaction intensity of the modeled fire. Fire behavior estimates are not reliable for conditions that produce an * in the display table.

3.8.2 Appearance of "*******"

The ****** indicates that the number displayed is larger than the allotted space for the Data General MV series computers. The number displayed is larger than 99999 when ****** appears. This sometimes occurs when you model severe conditions. This representation may differ for various other computers.

4. SAMPLE FIRECAST RUN

```
                    ******FIRECAST FIRE BEHAVIOR PROGRAM******
                           RIVERSIDE FIRE LABORATORY
                             JACK COHEN   APRIL 1985

DO YOU WANT HELP (Y/N)?
N
DESIGNATE THE OUTPUTS YOU DESIRE FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD
Y
FIRELINE INTENSITY
Y
FLAME LENGTH
Y
PERIMETER AND AREA
Y
SCORCH HEIGHT
Y
IGNITION COMPONENT
Y
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND, WIND INCREMENT
1,0,10,2
INPUT PERCENT SLOPE, TEMP, AND SKY COVER
30,80,0
INPUT FUEL MODEL SOURCE AND FUEL MODEL
SCAL,C
INPUT PERCENT 1-HR FM,10-HR FM, AND MONTH AND DAY
3,5,9,15
OPTIONAL USER INPUTS FOR SCAL
INPUT PERCENT DEAD AND BRUSH DEPTH (FT):
30,5
INPUT PERCENT NEW AND OLD GROWTH LIVE FUEL MOISTURES:
50,50
----------------------------------------------------------------------
                         SUMMARY OF INPUTS                           I
SLOPE:  30.   TEMP:  80.   SKY CVR: 0    1HR:  3.    10HR:  5.   I
DATE:  9/15   %DEAD:   30.  DEPTH:   5.   LIVE(NEW/OLD):  50./ 50.I
----------------------------------------------------------------------
SOUTHERN CALIFORNIA BRUSH FUEL MODEL
C...CHAMISE BRUSH
```

WIND AT MIDFLAME HEIGHT (MPH)

	0	2	4	6	8	10
RATE OF SPREAD (CH/HR)	5.	12.	24.	38.	53.	70.
FIRELINE INTENSITY (BTU/FT/SEC)	60.	160.	310.	488.	687.	902.
BYRAMS FLAME LENGTH (FT)	3.	5.	6.	8.	9.	10.
PERIMETER (CH/HR)	18.	43.	76.	109.	144.	179.
AREA (ACRES/SQ HR)	3.	14.	40.	75.	115.	155.
SCORCH HT (FT)	16.	30.	44.	54.	62.	67.
IGNITION COMP (PCT)	25.	41.	57.	71.	82.	82.

```
CONTINUE FIRECAST (Y/N)?
CHANGE OUTPUT DISPLAY (Y/N)?
Y

DESIGNATE THE OUTPUTS YOU DESIRE FOR THE SESSION (Y/N)
DO YOU WANT:
RATE OF SPREAD
Y
FIRELINE INTENSITY
Y
FLAME LENGTH
Y
PERIMETER AND AREA
N
SCORCH HEIGHT
N
IGNITION COMPONENT
Y
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
DISPLAY
              1                    0         10          2
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
2,0,20,5

INPUT PERCENT SLOPE, TEMP, AND SKY COVER
 (no data change, [NL])
INPUT FUEL MODEL SOURCE AND FUEL MODEL
 (no data change, [NL])
INPUT PERCENT 1-HR FM,10-HR FM, AND MONTH AND DAY
DISPLAY
            3       5           9      15
INPUT PERCENT 1-HR FM,10-HR FM, AND MONTH AND DAY
5,7,6,20
OPTIONAL USER INPUTS FOR SCAL
INPUT PERCENT DEAD AND BRUSH DEPTH (FT):
NO
```

```
INPUT BRUSH AGE (YRS):
45
INPUT PERCENT NEW AND OLD GROWTH LIVE FUEL MOISTURES:
100,80

-------------------------------------------------------------------
                        SUMMARY OF INPUTS                         I
SLOPE:  30.   TEMP:  80.    SKY CVR: 0   1HR:  5.    10HR:  7.    I
DATE:  6/20   AGE: 45.      LIVE(NEW/OLD): 100./ 80.              I
-------------------------------------------------------------------
SOUTHERN CALIFORNIA BRUSH FUEL MODEL
C...CHAMISE BRUSH
                       WIND AT 20 FT HEIGHT (MPH)
                         0       5       10      15      20
-------------------I-------I-------I-------I-------I-------I
RATE OF SPREAD     I       I       I       I       I       I
  (CH/HR)          I   4.  I  13.  I  26.  I  40.  I  56.  I
-------------------I-------I-------I-------I-------I-------I
FIRELINE INTENSITY I       I       I       I       I       I
  (BTU/FT/SEC)     I  66.  I 237.  I 464.  I 720.  I 995.  I
-------------------I-------I-------I-------I-------I-------I
BYRAMS FLAME LENGTHI       I       I       I       I       I
  (FT)             I   3.  I   6.  I   8.  I   9.  I  11.  I
-------------------I-------I-------I-------I-------I-------I
IGNITION COMP (PCT)I  16.  I  31.  I  43.  I  54.  I  60.  I
-------------------I-------I-------I-------I-------I-------I
CONTINUE FIRECAST (Y/N)?
Y
CHANGE OUTPUT DISPLAY (Y/N)?
N
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
DISPLAY
                 2                      0          20            5
INPUT 1=MIDFLAME OR 2=20FT HEIGHT, INITIAL WIND,FINAL WIND,WIND INCREMENT
2,0,30,5
INPUT PERCENT SLOPE, TEMP, AND SKY COVER
 (no data change, [NL])
INPUT FUEL MODEL SOURCE AND FUEL MODEL
NFFL,4
INPUT PERCENT 1HR,10HR,100HR AND LIVE
3,5,7,60

--------------------------------------------------------------------
                        SUMMARY OF INPUTS                          I
SLOPE:  30.   TEMP:  80.    SKY CVR: 0   1HR:  3.    10HR:  5.     I
100HR:  7.    LIVE:  60.                                           I
--------------------------------------------------------------------
NORTHERN FOREST FIRE LAB FUEL MODEL
4...MIXED BRUSH/HVY DEAD
```

	WIND AT 20 FT HEIGHT (MPH) 0	5	10	15	20	25
RATE OF SPREAD (CH/HR)	24.	69.	143.	236.	343.	461.
FIRELINE INTENSITY (BTU/FT/SEC)	1405.	3989.	8319.	13700.	19902.	26797.
BYRAMS FLAME LENGTH (FT)	13.	20.	29.	36.	43.	49.
IGNITION COMP (PCT)	29.	49.	71.	82.	82.	82.

CONTINUE FIRECAST (Y/N)?
N
**** STOP

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FIRECAST is a computer program that estimates fire behavior in terms of six fire parameters. Required inputs vary depending on the outputs desired by the fire manager. Fuel model options available to users are these: Northern Forest Fire Laboratory (NFFL), National Fire Danger Rating System (NFDRS), and southern California brushland (SCAL). The program has been operationally tested for use in the State of California. Complete operating instructions are given along with a description of program components and a complete sample run.

Retrieval Terms: fire behavior, fire modeling, fuel models